SOCIAL AND ECOLOGICAL DIMENSIONS OF PRAIRIE CONSERVATION: LINKING RANCHERS, RANGELAND HEALTH AND ABUNDANCE FOR THREE GRASSLAND SONGBIRD SPECIES AT RISK

A Thesis Submitted to the College of

Graduate Studies and Research

In Partial Fulfillment of the Requirements

For the Degree of Doctor of Philosophy

In the School of Environment and Sustainability

University of Saskatchewan

Saskatoon

Canada

By

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ABSTRACT

Temperate grasslands and the biodiversity they sustain are becoming increasingly imperilled. Habitat loss and degradation are considered primary causes of grassland species declines. Since livestock production is a dominant use of remaining temperate grassland, livestock producers and the grazing management decisions they make play a vital role in the recovery of grassland species. In this thesis, I examined social and ecological factors that drive habitat management and abundance of prairie wildlife species so as to contribute to conservation planning for prairie species at risk and their habitat. I focused on grassland songbirds because, of all prairie wildlife species, grassland birds have undergone some of the most dramatic declines in recent years. I employed an interdisciplinary approach, using theory and methodology from ornithology, rangeland management and the human dimension of conservation biology to achieve three objectives: i) to determine the extent to which indices of rangeland health explained variation in grassland songbird abundance for ten grassland bird species, including three species currently listed under Canada's Species at Risk Act: Sprague's pipit, McCown's longspur and Chestnut-collared longspur; ii) to describe livestock producer characteristics, summarize producer awareness of and attitudes towards species at risk and the Canadian Species at Risk Act and evaluate how characteristics, awareness and attitudes affect producer willingness to engage in voluntary stewardship actions that support species at risk conservation; and iii) to explore relationships between select social and ecological factors and bird abundance for the three aforementioned grassland bird species at risk to elucidate novel pathways for achieving their conservation. I address the first objective in Chapter 2, where I identify the rangeland health index as a poor predictor of bird abundance and vegetation structure variables, such as litter and vegetation volume, as strong predictors of bird abundance. These findings make a case for further refining the rangeland health index as a tool for biodiversity assessment. In Chapter 3 I achieve the second objective and summarize producer characteristics, awareness and attitudes towards species. I also identify awareness, attitudes and rangeland management learning approach as important to producer willingness to support species at risk recovery. I accomplish the third objective in Chapter 4, where I present results of a structural equation model that upholds bird-habitat relationships identified

in Chapter 2 and distinguishes management jurisdiction, size of land holdings and attitudes as important social factors to consider in conservation planning. Chapters 2 and 3 contribute to theory and methodology related to the ecological and social dimensions of grassland bird conservation, respectively. Chapter 4 demonstrates how structural equation models can be used to integrate social and ecological factors, and thereby inform habitat conservation and management. Both social and ecological data presented in this thesis make valuable contributions to producer engagement and habitat management aspects of conservation planning efforts for species at risk in the Milk River watershed of southwestern Saskatchewan. Overall, my findings point to the importance of a joint effort by regional private and public managers to use livestock grazing to create a mosaic of vegetation structure and habitat conditions suitable for the grassland bird community as a whole. This thesis provides a methodological approach that draws on and integrates social and ecological data, methods and concepts, thereby demonstrating how to conduct interdisciplinary research for biological conservation.

ACKNOWLEDGEMENTS

I sincerely thank my supervisors Dr. Maureen Reed and Dr. Stephen Davis and members of my supervisory committee, Dr. Ken Belcher, Dr. Jeff Thorpe, Dr. Keith Hobson, and Dr. Geoff Cunfer, for their continued support, mentorship and guidance. Thank you Dr. Cormack Gates, my external examiner, for challenging and insightful questions during my thesis defense. I'd also like to thank Dr. Doug Clark, Dr. Ryan Brook, Dr. Eric Lamb, and Pat Fargey for guidance during pivotal points of this process.

Thank you to the many people who assisted with data collection in the wilds of southwestern Saskatchewan in 2009 and 2010, including Holly Bogard, Ryan Clark, Ryan Dudrange, Nicholas Doane, Felicia Drever, Alicia Entem, Sean Fagan, Caroline Gaudet, Tiia Haapalainen, Val Harris, Michael Henry, Leah Kovatch, Jesse LaFramboise, Danielle Levesque, Susan Lutkin, Alex MacPhail, Remy Morriset and David Simpson. Thank you to Jesse La Framboise for returning for a second field season as a field crew leader and to Jesse La Framboise, Tiia Haapalainen, Nicholas Doane and Remy Morriset for their dedication and teamwork. A heartfelt thank you to Susan Lutkin for her perseverance, professionalism and creativity during interview data collection in the winter of 2009. Thank you Leanne Ejack and Danielle Levesque for attention to detail while working on data entry and quality.

Thank you to D. Sawatzky and J. Thorpe for field staff training, staff of Grasslands National Park for logistical support, Dr. Nicole Koper, Dr. Eric Lamb, Dr. Jill Johnstone, Dr. Carl Schwarz, and Steve VanWilgenburg for their statistical guidance, and Mark Gilchrist for GIS and mapping support.

A special thank you to the ranching families and pasture managers of southwestern SK for their hospitality and participation. Without their willingness to participate, none of this work would have been possible.

I'd like to thank the Government of Canada (Natural Sciences and Engineering Research Council of Canada, Environment Canada-Canadian Wildlife Service, and Species at Risk Inter-departmental Recovery Fund), Saskatchewan Fish and Wildlife Development Fund, University of

Saskatchewan, Nature Saskatchewan, the Canadian Federation of University Women and the Canadian Wildlife Federation for financial support that made this work possible.

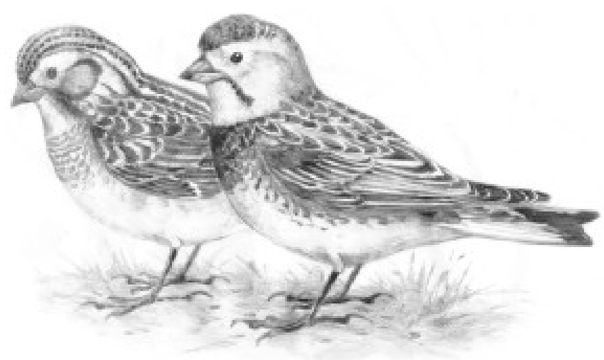
A heartfelt thank you to my community of family and friends whose encouragement and kindness carried me through the PhD process, especially Jean Kayira, Poornima Sheelanere, Asa Almstedt, Kristen Calverley, Kristen Gorman, Jeanine Bond, Tara Stephens, Pat Fargey, Anna Leighton, Susan Lutkin, Nathalie Brunet, Ross Phillips, Adam Minke, Clare Sullivan, Christine Stump, Ruby Stickney, Craille Maguire Gillies, Patricia Gillies, Jola Pisz, Alexis Harvey, Melissa Arcand, Faith Baron, Shannon Dyck, Colleen George, Franny Rawlyk, Jonathan Henderson, Heather Lake, Stephanie Henderson, Rylan Rogers, Laura Henderson-Shylo, Stephan Shylo, Kathleen Henderson, Ervin Henderson, Ewan Hunter and Garth Hunter.

Finally, a very special thank you to my guys, Garth and Ewan, for your patience and encouragement.

DEDICATION

I dedicate this thesis to Garth and Ewan, for your enduring patience and support,

and to all those striving to protect prairie species and habitats for future generations.



McCown's Longspur (*Rhynchophanes mccownii*) Illustration by Judie Short, Aurora Ontario (COSEWIC 2006).

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF ABBREVIATIONS

AICc	Akaike's Information Criterion
COSEWIC	Committee on the Status of Endangered Wildlife In Canada
ESA	Endangered Species Act
IUCN	International Union for the Conservation of Nature
PCAP	Prairie Conservation Action Plan

CHAPTER 1: INTRODUCTION- PRAIRIE CONSERVATION, LIVESTOCK PRODUCTION AND GRASSLAND BIRD HABITAT SELECTION

1.1 Introduction

Temperate grasslands and the biodiversity they sustain are becoming increasingly endangered. An estimated 41% of the world's native temperate grasslands and 79% of North American grasslands have been lost to cultivation for agricultural production (White *et al.* 2000). Remaining native grasslands are impacted by a growing human population, expanding food and fibre production, and extensive energy sector development. Habitat loss and degradation are considered primary causes of declines in grassland species around the world (White *et al.* 2000). North American grassland birds have declined sharply over the last four decades (Askins *et al.* 2007; Sauer *et al.* 2010) having undergone some of the most dramatic declines, showing steeper, more consistent declines across wider geographic areas than any other group of birds (Knopf 1996; Brennan and Kuvlesky 2005). Over the last decade, six of the nine endemic grassland songbird species in North America have declined to levels sufficient to warrant particular concern (Askins *et al.* 2007). Three species, Sprague's pipit (*Anthus spragueii*), Chestnut-collared longspur (*Calcarius ornatus*) and McCown's longspur (*Rhynchophanes mccownii*), are currently listed under Schedule 1 of Canada's Species at Risk Act as threatened (i.e., Sprague's pipit and Chestnut-collared longspur) and species of Special Concern (i.e., McCown's longspur) (Species at Risk Public registry 2013). Baird's sparrow (*Ammodramus bairdii*) and Lark bunting (*Calamospiza melanocorys*) (COSEWIC 2013) are of conservation concern and are candidate species for listing.

Declining avian population trends are believed to be strongly linked to habitat loss and degradation associated with changes in North American agricultural land use over the last century (Murphy 2003). Grassland bird habitat requirements are well-studied and are known to vary by species (Davis *et al.* 1999; Davis 2004; Fritcher *et al.* 2004; Smith and Lomolino 2004), many of which show distinct preferences for the structure and composition of the plant community (Madden *et al.* 2000; Fisher and Davis 2010). Bird species vary along a continuum of habitat features shaped by environmental conditions and disturbance such as grazing (Bock *et* *al.* 1993; Fritcher *et al.* 2004; Smith and Lomolino 2004; Fuhlendorf *et al.* 2006). Species such as Horned lark (*Eremophila alpestris*) and McCown's longspur are associated with pastures under relatively heavy grazing pressure whereas species such as Bobolink (*Dolichonyx oryzivorus*) and Sedge wren (*Cistothorus platensis*) are associated with lightly or ungrazed grasslands and Baird's sparrow and Sprague's pipit occupy grasslands with intermediate grazing pressure (Knopf 1996). Grassland generalist species, such as Vesper sparrow (*Pooecetes gramineus*) and Savannah sparrow (*Passerculus sandwichensis*) occupy a wider range of grassland habitats (Jones and Cornely 2002; Wheelwright and Rising 2008).

Livestock production is the predominant land-use on remaining native prairie (Tanaka *et al.* 2005b) and can enhance or degrade habitat via changes to the structure and function of rangeland plant communities (Fuhlendorf and Engle 2001). Cattle grazing may ultimately lead to a mosaic of grass species and habitat structure that varies across time and space (Turner and Chapin 2005; Romo 2007), thereby providing habitat for a wide variety of grassland species. Alternatively, it can create irreversible changes to rangeland and riparian plant communities, negatively impact ecosystem functioning and degrade habitat (Freilich *et al.* 2003). However, most research that examines relationships between aspects of livestock management and grassland bird habitat selection use vegetation measures that are bird-centric (Fisher and Davis 2010) and may be of little relevance to livestock producers or rangeland managers. For example, Robel (Robel *et al.* 1970) and Wiens pole (Wiens 1969) measurements, while useful in describing bird habitat structure, may have limited value to some rangeland managers and livestock producers.

Rangeland assessments based on vegetation measures have long been used in North America to determine the impact of grazing by cattle. While range condition methods were once predominantly used to evaluate the response of grassland vegetation to grazing (Dyksterhuis 1949), this methodology could not account for the wide spectrum of vegetation dynamics that occur on rangelands, including the irreversible impacts of invasive species and soil erosion (Task Group on Unity in Concepts and Terminology Committee 1995; Briske *et al.* 2005). For this reason, rangeland health indices were developed as a standard tool for assessing grassland structure and community composition and for indicating how close producers are to achieving optimal grassland health on a particular ecological site defined by soil and site stability, hydrologic function, and biotic integrity (Pyke *et al.* 2002; Adams *et al.* 2005; Pellant *et al.* 2005). These indices may be more useful for monitoring biodiversity than other methods of range assessment that are based primarily on plant species composition because some species may respond more strongly to vegetation structure. Although relationships between biodiversity and rangeland condition have been studied (Smith *et al.* 1995; Nelson *et al.* 1997; Bai *et al.* 2001; Fritcher *et al.* 2004), few studies examine the relationship between biodiversity and rangeland health (Bradford *et al.* 1998; Symstad and Jonas 2011). Rangeland health assessments differ from assessments based on rangeland condition in that rangeland health includes attributes of vegetation structure (Adams *et al.* 2005). If rangeland health indices can be used for biodiversity assessment, they may offer biologists and livestock producers a valuable tool to achieve grassland species conservation goals through grazing management. Chapter 2 of this thesis examines the feasibility of the rangeland health index as a grassland biodiversity assessment tool.

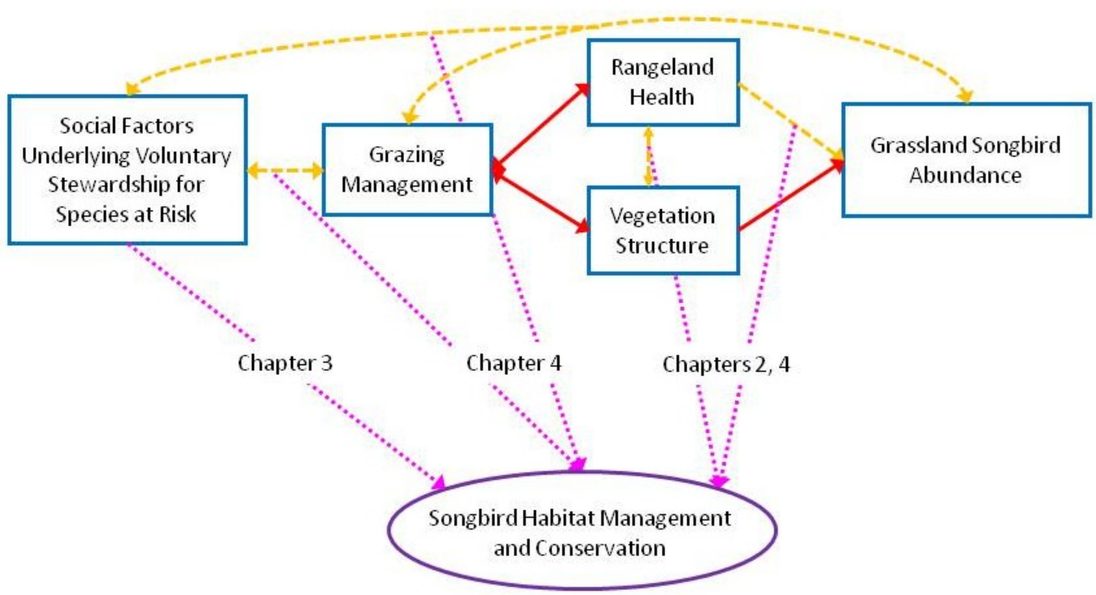
Given the predominance of livestock production on remaining native prairie and the capacity of cattle grazing to shape habitat, livestock producers and the grazing management decisions they make play a key role in the recovery of prairie species at risk on private lands. In Canada, the federal Species at Risk Act (2005) (SARA) supports voluntary stewardship as the preferred approach for conserving species at risk on private lands. Voluntary stewardship includes any voluntary action taken by a person to protect species at risk. There are several mechanisms to protect species, their residences or critical habitat under the SARA (Mooers *et al.* 2010). Once listed, all individuals and their residences receive protection on federal lands under the SARA's general prohibitions. On private lands, however, only individuals and residences of listed migratory birds or aquatic species are protected. Once a species' critical habitat has been identified by the federal government, the habitat receives immediate protection from destruction on federal lands or if it is considered aquatic. If provincial or territorial laws do not effectively protect the species' critical habitat or no such protection is in place, the federal government may apply the SARA's general prohibitions through an emergency order after consultation with affected stakeholders. In the case of an emergency order or where critical habitat has been identified on private lands, the Minister may provide compensation to private land managers for losses suffered. At the time of preparing this thesis, only one emergency order has been registered to come into effect Feb. 18, 2014 (for Greater Sage Grouse) (Her Majesty the Queen in Right of Canada 2013), however compensation for losses suffered on private lands where critical habitat has been identified has not yet been applied (Wojciechowski *et al.* 2011). Voluntary stewardship efforts for listed species on private lands are supported by the federal Habitat Stewardship Program, which provides funding and facilitates partnerships for activities that protect and conserve species at risk and their habitat. Few studies examine the protection of species at risk on private lands in Canada (e.g., Wojciechowski *et al.* 2011; Olive 2012) and equally few explore producer willingness to engage in voluntary stewardship of wildlife (Jacobson *et al.* 2003; Troy *et al.* 2005). For species at risk protection on private lands to be effective under the SARA, it is important to identify producer awareness of and attitudes towards species at risk and the factors that influence producers' willingness to protect species at risk on their lands. Chapter 3 of this thesis seeks to address this concern.

The research presented in this thesis is based on the premise that effective habitat protection relies on a thorough examination of both social and ecological factors in a system (Mascia *et al.* 2003), and more specifically, that interdisciplinary research is needed to integrate social and ecological science to improve Canadian species at risk recovery (Forester and Machlis 1996; Mace *et al.* 2001). Relationships between social and ecological dimensions of conservation problems are difficult to assess using singular disciplines and traditional approaches. Multivariate analyses offer conservation biologists a tool to examine multiple intercorrelated relationships between social and ecological variables (Grace 2006). In Chapter 4 of this thesis, I explore relationships between social and ecological variables that I postulate are important to the conservation and management of grassland songbirds.

1.2 Research Purpose and Approach

The research presented in this thesis seeks to fill gaps in knowledge surrounding recovery planning for grassland birds (Figure 1-1).The impacts of grazing management on rangeland health and on the vegetation structure that characterizes grassland bird habitat are well established in the literature (Bock *et al.* 1993*;* Derner *et al.* 2009), as are the effects of vegetation structure on grassland bird abundance (Fisher and Davis 2010) (Figure 1-1). What prairie conservation lacks is a tool to communicate species at risk habitat requirements with livestock producers (Chapter 2) and a clear understanding of the factors that influence the willingness of livestock producers to support species at risk (Chapter 3). Finally, although effective habitat protection requires a clear understanding of both social and ecological factors (Mascia *et al.* 2003), few examine relationships between social and ecological drivers of biodiversity loss (Forester and Machlis 2006; Mora 2008) (Chapter 4).

Figure 1-1. Concept map depicting relationships among social and ecological elements of the thesis research. Solid red arrows indicate relationships that are well-established in the literature; dashed yellow arrows indicate gaps in the literature that are examined in this thesis; dashed purple arrows indicate contributions of thesis chapters to grassland songbird habitat management and conservation.

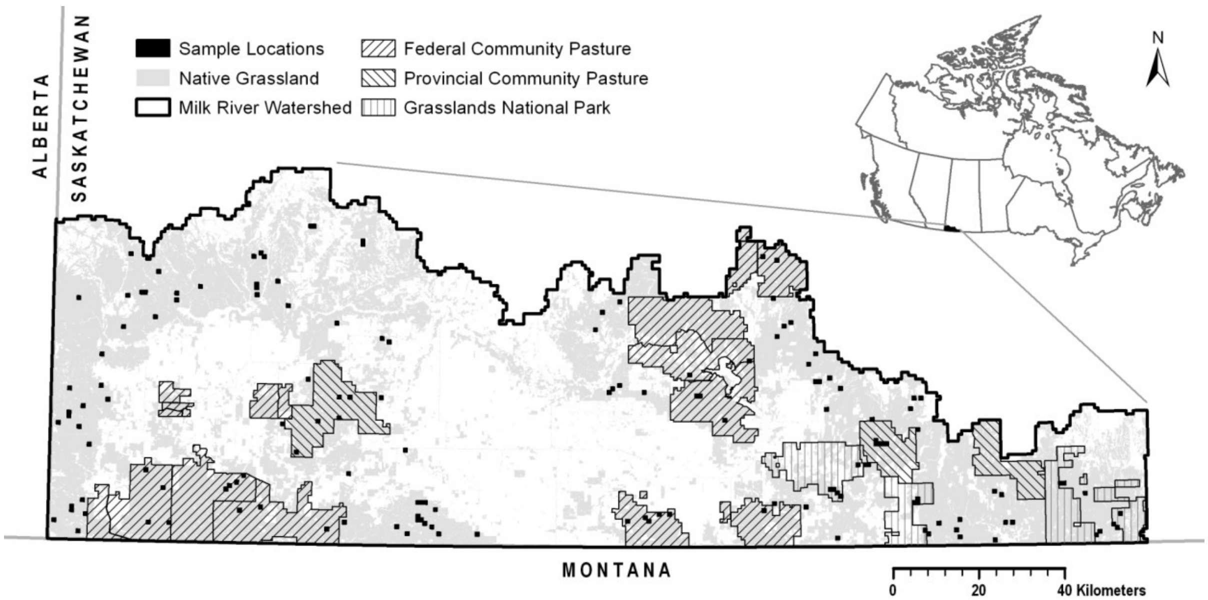


With an interdisciplinary approach, I sought to identify social and ecological variables of importance to the conservation of grassland songbird species considered at risk of extinction in Canada. I selected federal, provincial and private[1] grasslands of the Milk River watershed in southwestern Saskatchewan for this work as they represent some of the largest remaining intact and contiguous tracts of native prairie in Canada where livestock production is the primary land-use (Hammermeister *et al.* 2001)(Figure 1-2). I worked across a wide range of land ownership and management so as to best understand how a wide range of grazing practices shape habitat suitability for grassland birds. My objectives were to:

i. determine the extent to which indices of rangeland health explained variation in grassland songbird abundance for ten grassland bird species, including three species currently listed under the Species at Risk Act: Sprague's pipit, McCown's longspur and Chestnut-collared longspur;
ii. describe producer characteristics, summarize producer awareness of and attitudes towards species at risk and the Canadian Species at Risk Act and evaluate how characteristics, awareness and attitudes affect producer willingness to engage in voluntary stewardship actions that support species at risk conservation; and
iii. elucidate novel pathways for achieving grassland bird conservation by exploring relationships between select social and ecological factors and abundance for the three aforementioned species considered at risk (Figure 1-1).

[1] I refer to private land as that which is privately owned or privately managed through a crown lease.

Figure 1-2. Study area and locations of 140 quarter sections of native prairie sampled for bird abundance and vegetation in 2009 and 2010. Quarter sections were used to select interview participants in the Milk River watershed of south-western Saskatchewan, Canada.



Interdisciplinary scholarship is broadly described by some as the integration of disciplinary work and knowledge (Robinson 2008) and by others more specifically as a thorough integration and synthesis of multi-disciplinary theory and methodology from the beginning of a project (Mitchell 2002). I integrated methodology and theory from multiple disciplines at the outset of this research. My study design relied on theory and methodology from several fields of study: ornithology, rangeland ecology and the social dimension of wildlife conservation. In addition to contributing knowledge to each of these disciplines, this thesis also generates new interdisciplinary research questions relevant to prairie conservation (Chapter 5).

1.3 Thesis Structure

This thesis is presented in the 'dissertation by manuscript' style and follows the guidelines set out by the College of Graduate Studies and Research. Additionally, the research complied with all ethics and licensing requirements for the use of animal and human subjects as well as for conducting research on private and public lands. Following this introductory chapter, the thesis is organized into three manuscripts, each of which is presented as a single thesis chapter.

The first manuscript (Chapter 2), “Rangeland Health Assessment: A Useful Tool for Linking Range Management and Grassland Bird Conservation?” examines bird-habitat relationships for grassland songbirds, three of which were species at risk. Using an information theoretic approach and data from bird point count surveys, vegetation measurements and rangeland health assessments, I examine the support of three hypotheses explaining variation in bird abundance as a function of local vegetation measures: bird abundance is best explained by 1) vegetation structure, 2) vegetation structure heterogeneity, or 3) plant community. This Chapter demonstrates whether rangeland health index is a useful predictor of bird abundance and highlights how bird species compare in the niches they occupy along a gradient of rangeland health and associated vegetation characteristics.

The second manuscript (Chapter 3), “Voluntary Stewardship and the Canadian Species at Risk Act: Exploring Rancher Willingness to Support Species at Risk in the Canadian Prairies”, relies on a mixed-methods approach and data from 40 personal interviews with livestock producers. I summarize producer characteristics, explore producer awareness of and attitudes towards species at risk and identify how characteristics, awareness and attitudes influence producer willingness to support species at risk conservation.

The third manuscript (Chapter 4), “Modeling Social and Ecological Drivers of Abundance for Three Grassland Songbirds at Risk”, uses structural equation modelling to examine relationships between social and ecological drivers of abundance for three grassland birds considered species at risk: Sprague’s pipit, Chestnut-collared longspur and McCown’s longspur.

In closing, the final thesis chapter (Chapter 5), “Conclusion- Integrating Social and Ecological Science for Conservation“, presents a synopsis of significant findings from the three manuscripts that follow and then discusses contributions of these findings to grazing management and grassland bird conservation. I also highlight the challenges and opportunities of using an interdisciplinary approach. Finally, I recommend future research on specific topics related to rangeland management and grassland bird ecology and those more broadly associated with interdisciplinarity and prairie conservation.

This thesis includes appendices that provide additional information not included in the manuscripts submitted for publication. Appendix A includes supplemental information relevant to Chapter 2; figures include the quarter section sampling scheme, Cleveland dot-plots of bird abundance data and plots of the NMDS of plant species composition and tables include results of distance and removal bird point count sampling, and rangeland health parameter estimates and standard errors. Appendix B contains interview questions used to collect data presented in Chapter 3 and figures depicting trends in producer characteristics and generalized willingness to support species at risk. Permission to use or author rights from each publisher that allow use of the manuscripts in this thesis are included in Appendix C.

1.4 Copyright and Author Permissions

Chapters 2 through 4 of this thesis consist of manuscripts that are in press or are in preparation for submission. I provide the manuscript citations below in order to maintain consistency with copyright and author rights for each publisher. For all manuscripts, the student is the first author as per the College of Graduate Studies and Research guidelines for manuscript style theses.

Chapter 2: Henderson, A. E. and S. K. Davis. 2014. Rangeland Health Assessment: A Useful Tool for Linking Range Management and Grassland Bird Conservation? *Rangeland Ecology & Management.* 67:88-98. [Society for Range Management]

Chapter 3: Henderson, A.E., M. Reed and S. K. Davis. 2014. Voluntary Stewardship and the Canadian Species at Risk Act: Exploring Rancher Willingness to Support Species at Risk in the Canadian Prairies. *Human Dimensions of Wildlife: An International Journal*.19:17-32. [Taylor and Francis]

Chapter 4: Henderson, A. E., E. Lamb, S. K. Davis, and M. Reed. 2014. Modeling Social and Ecological Drivers of Abundance for Three Grassland Songbirds at Risk. *Conservation Biology*. Article in preparation for submission. [Wiley-Blackwell]

PREFACE TO CHAPTER 2: RANGELAND HEALTH ASSESSMENT: A USEFUL TOOL FOR LINKING RANGE MANAGEMENT AND GRASSLAND BIRD CONSERVATION?

Large-scale loss and degradation of North American native prairie coupled with sharp declines in grassland bird populations call for a clear understanding of the effects of livestock production on bird habitat selection. Grassland birds typically select breeding habitat based on a suite of structural and community vegetation features shaped by grazing. Rangeland health indices are a tool for assessing grassland structure and community composition that may offer biologists and range managers common language to achieve grassland bird recovery goals. Hence, the first objective of this thesis was to examine the feasibility of the rangeland health index as a tool for assessing biodiversity.

I used point count surveys, vegetation measures, and indices of rangeland health to examine bird-habitat relationships on native grassland in southwestern Saskatchewan for ten grassland bird species. I used an information theoretic approach to compare the support of three hypotheses explaining variation in bird abundance as a function of local vegetation characteristics: bird abundance is best explained by 1) vegetation structure, 2) vegetation structure heterogeneity, or 3) plant community. Vegetation structure variables were present in top-ranking models (i.e., models within 4 AICc units of top model) for eight species and solely comprised top-ranking models for Baird's sparrow *(Ammodramus bairdii)*, chestnut-collared longspur (*Calcarius ornatus*), horned lark (*Eremophila alpestris*), McCown's longspur (*Rhynchophanes mccownii*), and savannah sparrow (*Passerculus sandwichensis*). Structural heterogeneity variables were present in top-ranked models for grasshopper sparrow (*Ammodramus savannarum*), horned lark (*Eremophila alpestris*), and western meadowlark (*Sturnella neglecta*). Plant composition variables solely comprised top-ranking models for clay-colored sparrow (*Spizella pallida*) and were present in top-ranked models for grasshopper sparrow and vesper sparrow (*Pooecetes gramineus*). My results indicate that vegetation structure variables, namely litter mass, vegetation volume, and bare ground cover, best explain variation in bird abundance. Although the rangeland health index received little support as a predictor of bird abundance, vegetation structure components of the index could be used to communicate grazing management guidelines that maintain grassland bird habitat.

Chapter 2 is published in the journal Rangeland Ecology and Management. See: Henderson, A. E. and S. K. Davis. 2014. Rangeland Health Assessment: A Useful Tool for Linking Range Management and Grassland Bird Conservation? *Rangeland Ecology & Management*. 67:88-98.

CHAPTER 2: RANGELAND HEALTH ASSESSMENT: A USEFUL TOOL FOR LINKING RANGE MANAGEMENT AND GRASSLAND BIRD CONSERVATION?

2.1 Introduction

Temperate grasslands and the biodiversity they sustain are becoming increasingly endangered. An estimated 41% of the world's native temperate grasslands and 79% of North American grasslands have been lost to cultivation for agricultural production (White *et al.* 2000). Those that remain support a growing human population, expanding food and fibre production, and extensive energy sector development. Habitat loss and degradation are considered primary causes of grassland species declines world-wide (White *et al.* 2000). In North America, grassland birds have declined sharply over the last four decades (Askins *et al.* 2007; Sauer *et al.* 2010). Currently, 57 grassland wildlife species are considered at risk in North America, 28 of which are grassland birds (IUCN 2011). Since livestock production is a dominant use of remaining global temperate grassland (Samson and Knopf 1994; Ramankuty *et al.* 2008), grazing management plays a vital role in the recovery of grassland species.

Soils, climate, topography, and disturbance (i.e., fire, grazing and human land-use) shape grassland structure, function, and diversity, creating a mosaic of habitat patches across a landscape that is home to a variety of grassland birds (Wiens 1973; Fuhlendorf and Engle 2001; Askins *et al.* 2007). Grassland birds show distinct preferences for the structure and composition of the plant community (Madden *et al.* 2000; Fisher and Davis 2010). As a result, bird species assemblages vary along a continuum of habitat features shaped by environmental conditions and disturbance such as grazing (Bock *et al.* 1993; Fritcher *et al.* 2004; Smith and Lomolino 2004; Fuhlendorf *et al.* 2006). Species such as horned lark (*Eremophila alpestris*) and McCown's longspur (*Rhynchophanes mccownii*) are associated with pastures under relatively heavy grazing pressure whereas species such as bobolink and sedge wren are associated with lightly or ungrazed grasslands and Baird's sparrow (*Ammodramus bairdii*) and Sprague's pipit (*Anthus spragueii*) occupy grasslands with intermediate grazing pressure (Knopf 1996). Grassland generalist species, such as vesper sparrow (*Pooecetes gramineus*) and savannah sparrow

(*Passerculus sandwichensis*) occupy a wider range of grassland habitats (Jones and Cornely 2002; Wheelwright and Rising 2008).

Despite debate over the ecological merit of commercial livestock grazing (Savoury 1988; Jensen 2001; Freilich *et al.* 2003), grazing may be used to enhance grassland bird habitat, in part via changes in the structure and function of rangeland plant communities (Fuhlendorf and Engle 2001; Derner *et al.* 2009). Livestock grazing may ultimately lead to a mosaic of grass species and structure that varies across time and space (Turner and Chapin 2005; Romo 2007), thereby providing habitat for a wide variety of grassland birds. Most research attempting to uncover relationships between various aspects of livestock management and grassland bird habitat selection use bird-centric vegetation measures (Fisher and Davis 2010) that may be of little relevance to land managers. For example, Wiens pole measurements (Wiens 1969), while useful in describing bird habitat structure, may be difficult to relate to some rangeland managers and livestock producers in a meaningful way.

Rangeland assessments based on vegetation measures have long been used to determine the impact of grazing by cattle. While range condition methods were once predominantly used to evaluate the response of grassland vegetation to grazing (Dyksterhuis 1949), the methodology could not account for the wide spectrum of vegetation dynamics that occur on rangelands, including the irreversible impacts of invasive species and soil erosion (Task Group on Unity in Concepts and Terminology Committee 1995; Briske *et al.* 2005). Rangeland health[2] indices are a standard tool for assessing grassland structure and community composition and indicate how close producers are to achieving optimal grassland health on a particular ecological site defined by soil and site stability, hydrologic function, and biotic integrity (Pyke *et al.* 2002; Adams *et al.* 2005; Pellant *et al.* 2005). These indices may be more useful for monitoring biodiversity than other methods of range assessment based primarily on plant species composition because some species may respond more so to vegetation structure. Although relationships between biodiversity and rangeland condition have been studied (Smith

[2] The rangeland health index used in Saskatchewan and Alberta relies on visual estimates to score a series of questions that reflect five key indicators of rangeland health: plant species composition and structure, hydrologic function, nutrient cycling, site stability, and presence of invasive species.

et al. 1996; Nelson *et al.*1997; Bai *et al.* 2001; Fritcher *et al.* 2004), few studies have examined the relationship between biodiversity and rangeland health (Bradford *et al.* 1998; Symstad and Jonas 2011). If rangeland health indices can be used for biodiversity assessment, they may offer biologists and range managers a valuable tool to achieve grassland species conservation goals through grazing management.

My objective was to examine the feasibility of rangeland health as a rangeland assessment tool for biodiversity. I focus on 10 grassland bird species, including three species (Sprague's pipit, chestnut-collared longspur (*Calcarius ornatus*), and McCown's longspur) currently listed under Canada's Species at Risk Act (Species at Risk Public Registry 2013). Populations of these species are in decline across North America and ample research shows that livestock grazing (Bock *et al.* 1993) and vegetation structure (Fisher and Davis 2010) influence grassland songbird habitat selection. Furthermore, the abundance of some grassland songbirds has been used as an indicator of healthy prairie (Environment Canada 2008). I used an information theoretic approach to assess the support of three hypotheses explaining variation in bird abundance as a function of local vegetation measures: bird abundance is best explained by 1) vegetation structure, 2) vegetation structure heterogeneity, and 3) plant community. In doing so, I aimed to elucidate how bird species compare in the niches they occupy along a gradient of rangeland health and associated vegetation characteristics.

2.2 Methods

2.2.1 Study area and site selection

I selected the Milk River watershed of southwestern Saskatchewan, Canada as my study area as it contains the largest tracts of remaining native prairie grazed by livestock and the highest diversity of species at risk in the province (Figure 1-2). The region is comprised of mixed-grass and fescue prairie communities, largely dominated by *Elymus lanceolatus* (Scribn. and Sm.) Gould, *Pascopyrum smithii* (Rydb.) Barkworth and D.R. Dewey, *Calamagrostis montanensis* (Scribn.) Vasey, *Festuca hallii* (Vasey) Piper, *Festuca saximontana* Rydb., *Hesperostipa comata ssp. comata* (Trin. and Rupr.) Barkworth, or *Hesperostipa curtiseta* (Hitchc.) Barkworth (Thorpe 2007; ITIS 2013). Approximately 70%, 20%, and 10% of the native grassland in the study is under private, provincial, or federal management respectively. All grasslands in my study region were grazed by cattle, with the exception of Grasslands National Park. My random selection of 140 quarter sections captured a wide range of grazing practices typical for this region. While I did not quantify grazing intensity directly, it is reflected in the vegetation structure variables I measured.

I used ArcGIS 9.3 (Environmental Systems Research Institute 2008) to stratify my random sampling based on land management categories (federal, provincial and privately managed) and selected 140 quarter sections of upland native prairie as indicated by either loam or solonetzic soils. I used the quarter section (i.e., 160 acres or 65 ha) as my experimental unit because it offered a suitable area for rangeland health assessments and is the typical unit by which land is sold and managed in the province (McKercher and Wolfe 1986). I restricted my selection of quarter sections to those that were native rangeland, entirely within a single management jurisdiction, and were part of an upland grassland patch >145 ha to reduce any potential confounding effects of patch size on grassland songbird abundance (Johnson and Igl 2001; Davis 2004). I conducted bird and vegetation surveys on selected quarter sections in 2009 and 2010.

2.2.2 Bird abundance

I randomly positioned three point-count (Hutto *et al.* 1986) sampling stations 300 m apart from the centre of the point count and at least 100 m from edges within each quarter section (Figure A-1). In 2009 and 2010, trained surveyors conducted one 5-minute, 100-m radius point count at each survey location from 26 May to 3 July. Surveyors recorded singing males upon first detection inside and outside of the 100-m radius circle. Surveys took place from 0.5 hrs before sunrise until four hours after sunrise and during mornings with wind <20 kph and no precipitation. I constrained survey conditions to reduce variability in bird detection among counts (Rotella *et al.* 1999). I measured or estimated distance to each bird (Buckland *et al.* 2001) and recorded bird detections within three equal time periods during the survey period (Farnsworth *et al.* 2002). I used the sum of male birds aurally detected within 100 m over all three point counts as an index of abundance. I used only those individuals detected by song to estimate the number of territorial males breeding in each quarter section because I could not reliably separate females from non-singing males for most species.

2.2.3 Imperfect detection

Potential detection biases associated with point count surveys used to estimate bird abundance have undergone much criticism (Alldredge *et al.* 2007a, b, 2008; Efford and Dawson 2009; but see Johnson 2008). Inference based on bird counts adjusted for imperfect detection is considered an improvement over unadjusted counts (Buckland *et al.* 2001). Therefore, I used Distance (Buckland *et al.* 2001) and removal sampling (Farnsworth *et al.* 2002) to attempt to account for the probability of detecting a cue once it is given and the probability that a cue is given when the observer is present, respectively.

I used program Distance 6.0 Release 2 (Thomas *et al.* 2009) for species with >45 detections and modeled probability of detection for each species without covariates using the conventional distance sampling (CDS) engine and with observer and time of day as covariates using the multiple-covariate distance sampling (MCDS) engine (Marques *et al.* 2007). In both analyses I binned counts based on distance intervals deemed appropriate by previous studies (i.e., 0-20 m, 20-30 m, 30-40 m, 40-50 m, 50-75 m, 75-100 m) (Rotella *et al.* 1999). To remove outliers and facilitate model fitting, I right-truncated count data to 100 m for each species (i.e., $g(w) = 0.1$) (Buckland *et al.* 2001). I assessed model fit based on AICc value, Chi-square goodness of fit and visual assessment of detection and probability distribution functions.

I used R statistical software 2.14.1 (R Development Core Team 2011) and package RMARK 2.1.0 (Laake 2012) to estimate detection probabilities via removal sampling (Farnsworth *et al.* 2002). I examined the relationship between capture history and my treatment parameters of interest (e.g., rangeland health, litter, etc.). I fit closed-capture Huggins models (Huggins 1989) in RMARK using candidate models comprised of grouping variables (i.e., year, season) or individual covariates (i.e., minutes from sunrise, wind speed, cloud cover and observer). I selected the most parsimonious model based on AICc and goodness of fit (Burnham and Anderson 2002).

2.2.4 Vegetation assessment

Previous work suggests grassland bird habitat selection is mediated by grassland structure (e.g., litter, vegetation volume, bare ground cover), heterogeneity in structure, and plant community composition (Wiens 1974b; Rotenberry 1985; Fuhlendorf *et al.* 2006). I selected predictor variables from the Saskatchewan rangeland health index that I considered important for grassland birds, including litter mass (kg·ha-1), % cover of bare ground, club moss, lichen, and shrub, and proportional biomass of individual plant species. In addition, I selected vegetation features not directly included in the overall rangeland health index that may be important to grassland birds, including vegetation volume (Robel *et al.* 1970), plant species richness, and structural heterogeneity (Wiens and Rotenberry 1981; Fuhlendorf *et al.* 2006; Fisher and Davis 2010).

I collected all vegetation measurements in 24 plots (20 x 50 cm) distributed regularly along the lines between the three bird point count locations (Figure A-1). In each plot, I estimated the percentage that each plant species contributed to total plant biomass within the plot; I used this data to calculate species richness. I measured vegetation volume using a Robel pole with 2.5 cm increments (Toledo *et al.* 2008) and estimated 100% obscurity to the nearest cm in all cardinal directions. All measurements were assessed from 4 m away at a height of 1 m (Robel *et al.* 1970). I visually estimated litter mass (kg·ha-1) by hand raking to collect all dead plant material (e.g., standing stems, fallen stems, and leaf material and partially decomposed material) within the plot and compared this to a litter normal typical of the range site being evaluated (Adams *et al.* 2005; Saskatchewan PCAP Greencover Committee 2008). I visually estimated signs of erosion and percent cover of club moss (*Selaginella* spp.), lichen, and bare ground (i.e., any land surface not covered by vegetation). I measured shrub cover (%) using the line-intercept method (Canfield 1941) on three 100 m transects randomly located between point count centers.

Since plant species composition is an important component of the rangeland health index, I used Non-metric Multidimensional Scaling (NMS) in PC-ORD 4.0 (McCune and Mefford 1999) with a Sorenson distance measure and random start configuration to reduce the dimensionality of plant species composition (74 species) to one synthetic variable for use in further analyses

(McCune and Mefford 1999; Beals 2006). Ordination of the plant community largely partitioned plant species biogeographically into those typical of the Dry Mixed, Mixed Grassland and Cypress Upland ecoregions (Thorpe 2007), or invasive species (Figure A-2). Values from the single orthogonal axis of the NMS were used as a covariate in subsequent models.

2.2.5 Rangeland Health

I assigned each quarter section an index of rangeland health using the Saskatchewan Rangeland Health Index (Saskatchewan PCAP Greencover Committee 2008). Similar indices were developed in Alberta (Adams *et al.* 2005) and the USA (Pellant *et al.* 2005) and have been used in other rangeland studies (Desserud *et al.* 2010). I used site descriptions and visual guides outlined in the Saskatchewan Rangeland Health Index (Saskatchewan PCAP Greencover Committee 2008) to score a series of questions that reflect key indicators of rangeland health: plant species composition and structure, hydrologic function and nutrient cycling, site stability, and presence of invasive species. I assessed the plant species composition for each quarter section by comparing the proportional biomass of each species present to that of the reference community for that ecosite (Thorpe 2007). Plant communities that closely resembled the reference plant community received a "similarity index" score of 40 while those that showed minor, moderate, significant, or extreme alterations from the reference plant community received a score of 30, 15, 7, or 0, respectively. I visually assessed plant structure by examining the presence of low shrubs, tall graminoids and forbs, medium graminoids and forbs, and ground-covering graminoids, forbs, moss, and lichen. If plant layers closely resembled those of the reference community, I assigned a score of 10. If one, two, or three layers were absent, I assigned a score of 7, 3, or 0, respectively. I assessed the cover and density distribution of invasive species. If no invasive species were present, I assigned a score of 5 for both cover and density distribution. If invasive species coverage was ≤1%, or > 1%, I assigned a score of 3 or 0, respectively. I used a density distribution guide to rate the infestation of invasive species. Invasive species density distributions rated as low or high were assigned a score of 3 or 0, respectively. I visually assessed whether there was more soil erosion than expected for each site (e.g., hoof-shearing, pedestalling, etc.). Sites with no sign of soil movement beyond the natural extent for the site were assigned a score of 10. Sites with slight, moderate, and extreme amounts of soil movement were scored 7, 3, and 0, respectively. Sites where ≤10% of the area was exposed soil that was management-caused received a score of 5. Sites where 10 to 20%, 20 to 50%, and > 50% of their area was management-caused exposed soil received a score of 3, 2, or 0, respectively. In compliance with the rangeland assessment methodology, I visually assessed amounts and distribution of litter (kg·ha-1) as described above. Sites received a score of 25 if litter amounts were evenly distributed and 65 to 100% of amounts expected under moderate disturbance. Moderately patchy litter amounts in the range of 35 to 65% of the expected amount and greatly reduced litter with < 35% of the expected amount received scores of 13 and 0, respectively. I summed scores from all questions to determine the total rangeland health score (/100); this final score represented rangeland that can be broadly classified as "unhealthy" (<50), "healthy with problems" (50-75), or "healthy" (75-100).

2.2.6 Statistical Analysis

I used R statistical software 2.14.1 (R Development Core Team 2011) for all analyses. I checked for multi-collinearity and selected predictor variables that did not demonstrate strong correlation with each other ($r^2 \leq 0.5$). Predictor variables included litter mass, vegetation volume, bare ground cover, shrub cover, plant species richness, plant species composition, overall rangeland health score, similarity index and coefficients of variation for litter mass, vegetation volume, and bare ground cover. I did not include similarity index and overall rangeland health score in the same models because they were correlated with each other (r^2=0.7). All predictor variables were averaged to the quarter section and scaled by 0.01 to ensure model convergence.

I used an information theoretic approach (Burnham and Anderson 2002) to assess the support of three hypotheses explaining variation in bird abundance as a function of local vegetation measures; bird abundance is best explained by 1) vegetation structure, 2) vegetation structure heterogeneity, or 3) plant community. Models comprised combinations of either four structural variables (hereafter structure models; litter mass, bare ground cover, vegetation volume, and shrub cover), three variables associated with structural heterogeneity (hereafter heterogeneity models; coefficients of variation for litter mass, vegetation volume, and bare ground cover), or four community variables (hereafter community models; plant species richness, plant species composition, similarity index, and overall rangeland health score). My final suite of 35 candidate models included only main effects and subsets of all additive models for each suite of models (Table 2-1).

Given the high frequency of zero counts and over-dispersion in my bird abundance data (Figure A-3), I explored the fit of a suite of zero-inflated Poisson and negative binomial models against their non-inflated counterparts (Wenger and Freeman 2008). A binomial generalized linear model is used to model for species occurrence, while species abundance can be modeled by a Poisson or negative binomial distribution (Zurr *et al.* 2009). I assessed zero and non-inflated Poisson and negative binomial main effects models for the occurrence portion of the zero-inflated model using robel and litter because these variables are particularly important predictors of grassland bird habitat selection (Fisher and Davis 2010). Once I identified a suitable structural model for each species occurrence, I held this component constant and varied the abundance component for all species models.

Table 2-1. Candidate vegetation structure, vegetation structure heterogeneity and plant community models of abundance for 10 grassland bird species built on a priori hypotheses of grassland bird habitat selection; lit= litter mass (kg·ha-1), robel= vegetation volume (cm^3), bg= bare ground cover (%), shrub= shrub cover (%), cvlit= litter mass heterogeneity, cvrob= vegetation volume heterogeneity, cvbg= bare ground heterogeneity, comp= plant species composition, sim= similarity index, rich= plant species richness, and rh= rangeland health score.

Structure	Heterogeneity	Community
lit + robel + bg + shrub	cvlit + cvrob + cvbg	comp + sim + rich
lit + robel + bg	cvlit + cvrob + cvbg	comp + rich + rh
robel + bg+ shrub	cvlit + cvrob	comp + sim
lit + bg + shrub	cvlit + cvbg	sim + rich
lit + robel +shrub	cvlit + cvrob	rich + rh
lit + robel	cvlit + cvbg	comp + rich
robel + bg	cvrob + cvbg	comp + rh
bg + shrub	cvlit	sim
lit + bg	cvrob	rich
lit + shrub	cvbg	rh
robel + shrub		
robel		
bg		
shrub		
lit		

Models were ranked using Akaike's Information Criterion adjusted for small sample size (AICc) (Anderson *et al.* 2001; Anderson 2008) and selected using Chi-square goodness of fit (Burnham and Anderson 2002). I examined residual plots to ensure that I met assumptions associated with generalized linear models. I addressed model selection uncertainty and effects of uninformative parameters using a model-averaging approach (Burnham and Anderson 2002; Arnold 2010). I model-averaged all variables within 4 AICc units of the top model and calculated their relative variable importance values (Burnham and Anderson 2002) using the MuMIn R package (Barton 2012). I chose to model-average all variables within 4 AICc units of the top model because this allowed for variable cumulative model probabilities or weights (w_i) to sum to approximately 0.90, and including models with lower weights would have little effect on the parameter estimates (Burnham and Anderson 2002). Although I calculated relative variable importance values across an unequal number of models, this did not affect cumulative weights for top-ranked variables because variables in models ranked below the top-ranking models had extremely small weights. *Post hoc*, I combined top-ranked structure, community and heterogeneity models to explore whether a combination of *a priori* hypotheses ultimately improved model fit. I considered a variable an important predictor of bird abundance if the 85% confidence interval did not include zero (Arnold 2010).

2.3 Results

2.3.1 Imperfect detection

I had sufficient detections to analyze bird habitat relationships for the following species: Baird's sparrow, chestnut-collared longspur, clay-colored sparrow (*Spizella pallida*), grasshopper sparrow (*Ammodramus savannarum*), horned lark, McCown's longspur, savannah sparrow, Sprague's pipit, vesper sparrow, and western meadowlark (*Sturnella neglecta*).

Few birds detected close to 0 m from the observer for all species yielded poor goodness of fit and a problematic shape in the detection function for all distance sampling models. According to assumptions of Distance, the shape of detection probability functions should demonstrate that detection probability decreases as distance from the observer increases (Buckland *et al.* 2001). However, in the detection probability functions generated from my data, few birds were detected near 0m and detection increased at approximately 20 m (Figure A-4). It is not known whether few birds detected near 0 m were due to evasive movement of birds away from the observer or a lack of bird response close to 0 m. Regardless, these results violated the assumption that all birds at 0 m are detected with certainty (i.e., *g(0)*=1) (Buckland *et al.* 2001) (Figure A-4). Therefore, I did not adjust my data to account for potential detection error associated with distance from the observer.

My removal sampling results showed that more birds were detected in the first interval than the second and third intervals for all species. Examination of relationships between variables related to capture history (wind speed, observer, minutes from sunrise, and cloud cover) and treatments (rangeland health, vegetation volume, litter, and species richness) yielded no outliers or patterns to warrant adjusting counts. Only abundance of chestnut-collared longspur and Sprague's pipit held potential for adjustment to account for detection probability associated with minutes from sunrise and cloud cover, respectively. However, I did not adjust my counts because it would result only in a scaling up or down of abundance without any meaningful consequences for the relationship between bird abundance and my explanatory variables. Abundance data for all other species did not require adjustment due to lack of model fit; either the null model was the best-supported model, model weights were consistently low, or confidence intervals for estimates of detection probability included zero (Anderson 2008) (Table A-1).

2.3.2 Bird–vegetation relationships

For the occurrence component of the zero-inflated model, the best-supported model for Baird's sparrow, chestnut-collared longspur, horned lark, McCown's longspur, savannah sparrow, and Sprague's pipit occurrence was a zero-inflated negative binomial (ZINB) model with vegetation volume and litter mass as covariates (Table 2-2). For Baird's sparrow, savannah sparrow, and Sprague's pipit the probability of occurrence increased with greater vegetation volume and litter mass while the opposite was found for chestnut-collared longspur, horned lark, and McCown's longspur. The best-supported model of occurrence for clay-colored sparrow and grasshopper sparrow was a ZINB model with vegetation volume as a covariate; the probability of occurrence for both species increased with increasing vegetation volume. The top occurrence model for vesper sparrow and western meadowlark was a ZINB model with no covariates assigned and vegetation volume, respectively. Confidence intervals for estimates of abundance of vesper sparrow and western meadowlark across all occurrence models overlapped zero.

Vegetation structure variables were present in top-ranking models (i.e., models within 4 AICc units of top model) for 8 species and solely comprised top-ranking models for 5 of 10 species (Baird's sparrow, chestnut-collared longspur, McCown's longspur, savannah sparrow, and Sprague's pipit; Table 2-2). Litter mass was an important predictor of Baird's sparrow, chestnut-collared longspur, horned lark, McCown's longspur, and savannah sparrow abundance (Table 2-3). Baird's sparrow and savannah sparrow abundance increased with litter mass, whereas chestnut-collared longspur, horned lark, and McCown's longspur abundance decreased (Figure 2-1A). Vegetation volume was an important predictor of Baird's sparrow, chestnut-collared longspur, grasshopper sparrow, horned lark, McCown's longspur, and Sprague's pipit abundance (Table 2-3). Abundance of Baird's sparrow, grasshopper sparrow, and Sprague's pipit increased with vegetation volume while chestnut-collared longspur, horned lark, and McCown's longspur abundance decreased. Baird's sparrow abundance increased sharply from 0.05 to 0.15 cm then rose slowly, whereas McCown's longspur abundance decreased sharply from 0.05 to 0.2 cm and declined slowly thereafter (Figure 2-1B). Bare ground cover explained variation in abundance for Baird's sparrow, horned lark, McCown's longspur, and savannah sparrow (Tables 2 and 3). Abundance of McCown's longspur and horned lark steadily increased with bare ground cover whereas Baird's sparrow and savannah sparrow abundance steadily decreased (Figure 2-1C). Shrub cover influenced the abundance of clay-colored sparrow, grasshopper sparrow, McCown's longspur, and savannah sparrow but its effect was more variable than other structural covariates. Abundance of clay-colored, grasshopper, and savannah sparrows increased with shrub whereas McCown's longspur abundance decreased (Table 2-3).

Table 2-2. Final ranking of candidate models relating bird abundance to vegetation characteristics for 10 grassland bird species on 140 quarter sections in southwestern Saskatchewan. Only those models with AICc values lower than the null model are presented. Species common name, model rank and structure, number of model parameters (K), log likelihood values (log(L)), AICc values of the null model and top-ranked models (ΔAICc<4), delta AICc values (Δi) and AICc weights (wi) are presented; lit= litter mass (kg·ha-1), robel= vegetation volume (cm^3), bg= bare ground cover (%), shrub= shrub cover (%), cvlit= heterogeneity of litter mass, cvrob= heterogeneity of vegetation volume, cvbg= heterogeneity of bare ground cover, comp= plant species composition, sim= similarity index, rich= plant species richness, and rh= rangeland health score. Zero-inflated models are presented as abundance component | occurrence component. Asterisks indicate the null model.

Species	Rank	Model Structure	K	log(L)	AIC_c	Δ_i	w_i
Baird's sparrow	1	lit + bg + robel| robel + lit	7	-525.1	1066.8	0.0	0.27
	2	lit + bg |robel + lit	6	-526.3	1067.1	0.3	0.23
	3	lit + robel + bg + shrub |robel + lit	8	-524.3	1067.3	0.5	0.21
	4	lit + bg + shrub |robel + lit	7	-525.8	1068.3	1.5	0.13
	5	lit + robel |robel + lit	6	-527.3	1069.0	2.2	0.09
	6	robel + bg |robel + lit	7	-526.4	1069.4	2.6	0.07
	23	1 |robel + lit *	4	-535.2	1080.7	13.9	0.0
chestnut-collared	1	lit + robel |robel + lit	6	-652.4	1319.3	0.0	0.37
	2	lit + robel + shrub |robel + lit	7	-651.9	1320.5	1.1	0.21
	3	lit + robel + bg |robel + lit	7	-652.4	1321.4	2.0	0.14

longspur	4	lit |robel + lit	5	-654.9	1322.1	2.8	0.09
	5	lit + robel + bg + shrub |robel + lit	8	-651.9	1322.5	3.2	0.07
	6	lit + bg |robel + lit	6	-654.4	1323.3	4.0	0.05
	27	1 |robel + lit *	4	-665.1	1340.5	21.2	0.0
	1	rich |robel	4	-163.4	337.1	0.0	0.31
	2	rich + rh | robel	5	-163.3	339.0	1.9	0.12
clay-colored	3	comp + rich | robel	5	-163.3	339.0	1.9	0.12
sparrow	4	sim + rich | robel	5	-163.4	339.2	2.1	0.11
	5	shrub | robel	4	-165.3	340.9	3.8	0.05
	6	comp + rich + rh |robel	6	-163.3	341.1	3.9	0.04
	12	1 |robel *	3	-167.4	343.0	5.9	0.0
	1	sim + rich | robel	5	-107.7	227.8	0.0	0.21
	2	rich | robel	4	-109.6	229.5	1.7	0.09
	3	comp + sim + rich | robel	6	-107.7	229.8	2.0	0.08
grasshopper	4	cvlit+ cvrob | robel	5	-108.8	229.9	2.1	0.07
sparrow	5	cvlit + cvbg | robel	5	-108.9	230.1	2.3	0.07
	6	cvbg | robel	4	-110.2	230.7	2.9	0.05
	7	rich + rh | robel	5	-109.2	230.7	2.9	0.05
	8	cvlit + cvrob + cvbg | robel	6	-108.1	230.7	2.9	0.05
	9	robel + shrub | robel	5	-109.3	231.0	3.2	0.04

	10	cvrob + cvbg |robel	5	-109.3	231.0	3.2	0.04
	11	cvrob | robel	4	-110.5	231.3	3.5	0.04
	12	comp + rich | robel	5	-109.6	231.6	3.8	0.03
	21	1 |robel *	3	-113.0	234.2	6.4	0.0
horned lark	1	lit + robel + bg | robel + lit	7	-562.1	1140.7	0.0	0.23
	2	cvbg | robel + lit	5	-564.4	1141.1	0.4	0.19
	3	cvrob + cvbg | robel + lit	6	-563.6	1141.6	0.9	0.15
	4	lit + robel | robel + lit	6	-564.1	1142.7	2.0	0.09
	5	lit + robel + bg + shrub | robel + lit	8	-562.1	1142.9	2.1	0.08
	6	cvlit + cvbg | robel + lit	6	-564.3	1143.0	2.3	0.07
	7	robel + bg | robel + lit	6	-564.5	1143.4	2.7	0.06
	8	cvlit + cvrob + cvbg | robel + lit	7	-563.5	1143.6	2.9	0.06
	24	1 |robel + lit *	4	-578.3	1167.0	26.2	0.0
McCown's longspur	1	lit + robel + bg + shrub | robel + lit	8	-332.2	683.2	0.0	0.34
	2	lit + robel + shrub| robel + lit	7	-334.0	684.6	1.4	0.17
	3	robel + bg + shrub | robel + lit	7	-334.3	685.2	2.0	0.12
	4	lit + robel + bg | robel + lit	7	-334.4	685.4	2.1	0.12
	5	lit + robel | robel + lit	6	-335.7	685.9	2.7	0.09
	6	robel + shrub | robel + lit	6	-335.7	686.0	2.7	0.09
	29	1 |robel + lit *	4	-347.3	704.7	21.5	0.0

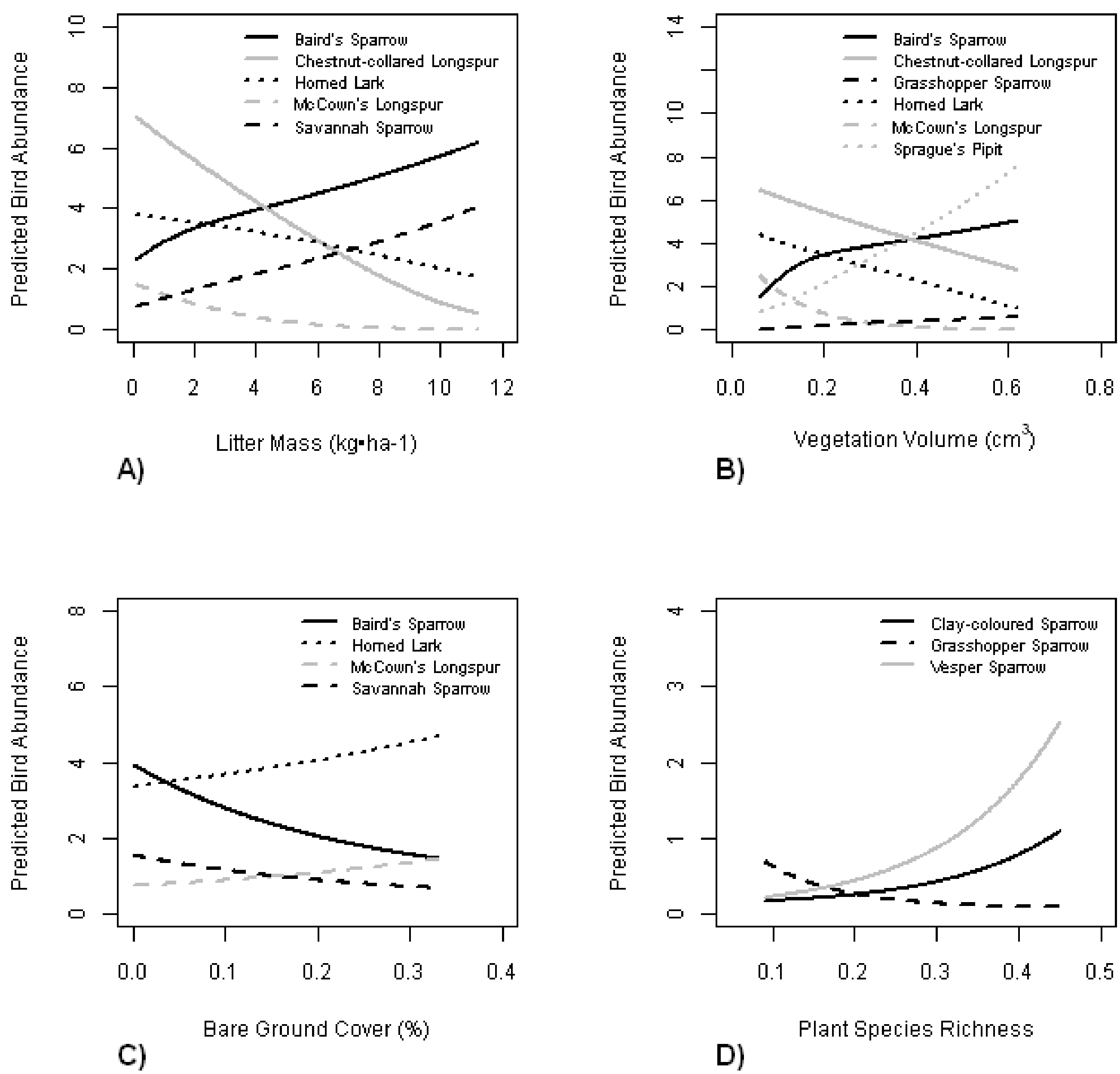
	1	lit + bg + shrub| robel + lit	7	-375.4	767.3	0.0	0.39
savannah sparrow	2	lit + bg | robel + lit	6	-377.5	769.5	2.1	0.13
	3	lit + robel + bg + shrub | robel + lit	8	-375.4	769.5	2.1	0.13
	4	lit + shrub| robel + lit	6	-377.6	769.6	2.3	0.12
	29	1 |robel + lit *	4	-392.0	794.3	27.0	0.0
	1	robel | robel + lit	5	-433.5	879.4	0.0	0.31
	2	robel + shrub | robel + lit	6	-433.1	880.7	1.3	0.16
	3	robel + bg | robel + lit	6	-433.3	881.0	1.7	0.14
Sprague's pipit	4	lit + robel | robel + lit	6	-433.5	881.4	2.0	0.12
	5	robel + bg + shrub | robel + lit	7	-432.9	882.4	3.0	0.07
	6	lit + robel + shrub | robel + lit	7	-433.0	882.7	3.3	0.06
	7	lit + robel + bg | robel + lit	7	-433.1	882.8	3.4	0.06
	16	1 |robel + lit*	4	-439.5	889.3	9.9	0.0
	1	comp + rich | 1	4	-303.0	616.3	0.0	0.52
vesper sparrow	2	comp + rich + rh | 1	5	-302.9	618.1	1.8	0.21
	3	comp + sim + rich | 1	5	-302.9	618.2	1.9	0.20
	17	1 |1*	2	-311.1	628.4	12.1	0.0
	1	cvlit |robel	4	-213.8	437.9	0.0	0.16
western meadowlark	2	cvlit + cvrob | robel	5	-213.1	438.5	0.6	0.12
	4	1 |robel *	3	-215.9	439.9	2.1	0.1

Table 2-3. Model-averaged parameter estimates and relative variable importance values calculated over top-ranking models (Δ AICc ≤4) for abundance of 10 grassland bird species in southwestern Saskatchewan. Asterisks indicate model parameter estimates with 85% confidence limits that do not include zero. Variable rank and name, cumulative weight of variable over top models (w+), model averaged parameter estimate (βe), and unconditional standard error (U.SE) are presented; lit= litter mass (kg·ha-1), robel= vegetation volume (cm^3), bg= bare ground cover (%), shrub= shrub cover (%), cvlit= heterogeneity of litter mass, cvrob= heterogeneity of vegetation volume, cvbg= heterogeneity of bare ground cover, comp= plant species composition, sim= similarity index, rich= plant species richness, and rh= rangeland health score.

Species	Rank	Variable	w_+	β^e	U.SE
	1	lit	1.00	0.072*	0.03
Baird's	2	bg	0.84	-4.274*	1.93
sparrow	3	robel	0.64	1.198*	0.70
	4	shrub	0.41	-3.870	3.24
		Intercept	---	1.156*	0.23
	1	lit	1.00	-0.120*	0.03
chestnut-	2	robel	0.85	-1.354*	0.62
collared	3	shrub	0.30	2.861	2.89
longspur	4	bg	0.28	0.478	1.09
		Intercept	---	2.181	2.89
	1	rich	0.94	7.387*	2.77
	2	comp	0.22	-7.600	19.59
clay-colored	3	rh	0.22	-0.458	1.15
sparrow	4	sim	0.15	-0.239	1.25
	5	shrub	0.06	14.207*	7.42
		Intercept	---	-2.771*	1.04
	1	rich	0.56	-10.800*	4.05
	2	sim	0.35	-2.924*	1.50
grasshopper	3	cvbg	0.25	0.319*	0.18
sparrow	4	cvrob	0.25	-3.315*	1.99
	5	cvlit	0.23	-1.190*	0.68

	6	comp	0.13	4.242	25.37
	7	rh	0.06	1.254	1.35
	8	robel	0.05	3.058*	1.13
	9	shrub	0.05	28.129*	13.89
		Intercept	---	2.057	2.28
horned lark	1	cvbg	0.51	-0.349*	0.07
	2	robel	0.49	-2.479*	0.74
	3	lit	0.43	-0.067*	0.03
	4	bg	0.40	2.130*	1.04
	5	cvrob	0.22	0.585	0.46
	6	cvlit	0.14	-0.53*	0.14
	7	shrub	0.09	0.299	3.26
		Intercept	---	1.860*	0.23
McCown's longspur	1	robel	1.00	-6.876*	2.09
	2	shrub	0.78	-11.684*	5.92
	3	lit	0.77	-0.279*	0.15
	4	bg	0.63	2.982*	1.80
		Intercept	---	2.063*	0.34
savannah sparrow	1	lit	1.00	0.115*	0.04
	2	bg	0.84	-4.324*	2.21
	3	shrub	0.83	6.697*	3.22
	4	robel	0.17	-0.053	0.88
		Intercept	---	0.320*	0.20
Sprague's pipit	1	robel	1.00	2.177*	0.67
	2	shrub	0.32	-3.145	3.51
	3	bg	0.29	-1.828	2.54
	4	lit	0.25	-0.16*	0.03
		Intercept	---	0.761*	0.18
vesper sparrow	1	comp	1.00	-43.977*	14.95
	2	rich	1.00	7.322*	1.96
	3	rh	0.22	-0.354	0.70
	4	sim	0.21	0.307	0.77
		Intercept	---	-2.332*	0.70
western meadowlark	1	cvlit	1.00	0.727*	0.37
	2	cvrob	0.35	-1.406	1.16
	3	cvbg	0.18	-0.053	0.15
		Intercept	---	-1.259*	0.66

Figure 2-1. Relationships between predicted bird abundance and covariates where 85% confidence intervals do not include zero: A) litter mass (kg·ha-1), B) vegetation volume (cm^3), C) bare ground cover (%), and D) plant species richness for nine species of grassland birds. Each x-axis represents the range of values recorded in the field, scaled by 0.01 to ensure model convergence. Model averaged estimates and unconditional standard errors for relationships are reported in Table 2-3.



Species richness, species composition, similarity index or the overall rangeland health index were present in top-ranking models for only clay-colored, grasshopper, and vesper sparrows and heterogeneity variables were present in top models for only grasshopper sparrow, horned lark and western meadowlark (Table 2-2). Rangeland health was not a strong predictor of bird abundance as confidence intervals overlapped zero for all species (Table A-2). Model averaged estimates of abundance indicated that plant species richness or heterogeneity of vegetation structure were important predictors of abundance for clay-colored sparrow, grasshopper sparrow, horned lark, vesper sparrow, and western meadowlark (Table 2-3). Species richness was an important predictor of abundance for clay-colored, grasshopper, and vesper sparrows (Table 2-3); clay-colored sparrow and vesper sparrow abundance increased with species richness whereas grasshopper sparrow decreased (Figure 2-1D). Grasshopper sparrow abundance decreased on rangeland characterised by greater heterogeneity of vegetation structure in general (Table 2-3). Horned lark abundance was greatest in areas characterised by lower heterogeneity of bare ground cover and western meadowlark abundance was greatest in areas characterised by increasing heterogeneity of litter (Table 2-3).

Combining top models from each hypothesis (i.e., vegetation structure, structural heterogeneity, and plant community) provided little improvement in model fit across species (Table 2-4). Combined models were within 2 AICc units of the best single model for all but vesper sparrow and horned lark. Model fit was improved for these species by the addition of vegetation structure variables; vegetation volume (ΔAICc= -3.8) and heterogeneity of bare ground cover (ΔAICc= -4.0), respectively.

Table 2-4. Results from combining top models (ΔAICc <4) from structure, heterogeneity and community hypotheses for 10 grassland bird species in southwestern Saskatchewan. Log likelihood (log(L)), AICc, ΔAICc, model averaged parameter estimates (βe) and unconditional standard errors (U.SE) are presented; lit= litter mass (kg·ha-1), robel= vegetation volume (cm^3), bg= bare ground cover (%), shrub= shrub cover (%), cvlit= heterogeneity of litter mass, cvrob= heterogeneity of vegetation volume, cvbg= heterogeneity of bare ground cover, comp= plant species composition, sim= similarity index, rich= plant species richness, and rh= overall rangeland health score. Asterisks indicate model parameters with 85% confidence limits that do not include zero.

Species	Model Structure	log(L)	AIC_c	Δ AIC_c	Variable	β^e	U.SE
Baird's sparrow	lit + bg + robel + rich + sim | robel + lit	-521.75	1064.40	2.40	lit	0.060*	0.03
					bg	-4.229*	1.91
					robel	1.010*	0.69
					rich	1.592*	0.95
					sim	-0.883*	0.43
chestnut-collared longspur	lit + robel + cvbg + rich | robel + lit	-649.85	1318.40	0.94	lit	-0.991*	0.22
					robel	-0.741	0.68
					cvbg	-0.107*	0.07
					rich	-1.194*	0.76
clay-colored sparrow	rich + shrub | robel	-162.29	336.90	0.24	rich	6.144*	2.52
					shrub	10.815*	7.40
grasshopper sparrow	robel + shrub + sim + rich + cvlit + cvrob | robel	-104.11	229.10	0.00	robel	1.857	1.70
					shrub	34.055*	19.79
					sim	-2.838*	1.30
					rich	-9.002*	3.59
					cvlit	-0.267	0.85
					cvrob	-0.532	2.36

horned lark	lit + robel + bg + cvbg | robel + lit	-560.05	1136.70	4.02	lit	-0.075*	0.03
					robel	-1.764*	0.74
					bg	0.749	1.13
					cvbg	-0.234*	0.08
McCown's longspur	lit + robel + bg + shrub + cvrob | robel + lit	-329.67	680.30	2.30	lit	-0.310*	0.14
					robel	-5.765*	1.99
					bg	3.140*	1.81
					shrub	-14.305*	6.09
					cvrob	2.401*	1.07
savannah sparrow	lit + bg + shrub + rich + rh | robel + lit	-372.03	765.00	2.34	lit	0.079*	0.04
					bg	-3.801*	2.18
					shrub	5.664*	3.19
					rich	2.940*	1.15
Sprague's pipit	robel + rich + sim | robel + lit	-430.61	877.80	1.59	robel	2.160*	0.65
					rich	1.041	0.97
					sim	-0.933*	0.44
vesper sparrow	comp + rich + lit | 1	-300.00	612.40	3.87	comp	-31.376*	15.03
					rich	8.019*	1.96
					lit	-0.154*	0.06
western meadowlark	NA[3]	NA[2]	NA[2]	NA[2]	NA[2]	NA[2]	NA[2]

[3] A combined hypothesis model was not run for western meadowlark because only heterogeneity variables outperformed the null model for this species.

2.4 Discussion

My findings are consistent with previous research identifying vegetation structure as an important predictor of grassland bird abundance. I demonstrate that while rangeland health itself is not a strong predictor of bird abundance, structural components used to assign this index are. Since abundance varies with vegetation features by bird species, my results provide further evidence that landscape-level spatial heterogeneity in vegetation structure is necessary for the conservation and recovery of the grassland bird community (Samson and Knopf 1996; Fuhlendorf *et al.* 2006; Askins *et al.* 2007). Widespread adoption of livestock production strategies that maximize economic gains through moderate grazing (Ritten *et al.* 2010) may reduce heterogeneity of vegetation structure and thereby limit the amount of available habitat for a number of species (Derner *et al.* 2009, Toombs and Roberts 2009). A mosaic of vegetation structure on rangelands is necessary to maximize grassland bird diversity and abundance at both the local and landscape scale (Madden *et al.* 2000; Fritcher *et al.* 2004; Fuhlendorf *et al.* 2006).

The rangeland health index received little support as a predictor of bird abundance. Although litter mass and bare ground are important to grassland birds (Fisher and Davis 2010), the categories used to estimate structural components of the rangeland health index may be too broad to explain variation in bird abundance. For example, my measures of litter that ranged from 8 kg·ha-1 to 1121 kg·ha-1 were assigned to three categories for the rangeland health index (Saskatchewan PCAP Greencover Committee 2008). While these categories may be useful for assessing rangelands, they may be too coarse to adequately relate bird abundance to rangeland health. Furthermore, shrub cover and density receive only 10 points in the rangeland health index and vegetation volume is not explicitly considered, yet both are known predictors of grassland bird abundance (Fisher and Davis 2010). Finally, plant community composition comprises 60% of the rangeland health index but it is not as important to bird habitat selection as vegetation structure (Fisher and Davis 2010; this study), which is allotted 40%. High rangeland health values are thought to represent improved ecological processes and better habitat quality for wildlife (Saskatchewan PCAP Greencover Committee 2008). This may only hold true for wildlife species whose habitat requirements are more closely linked to plant species composition than vegetation structure, such as shrubsteppe birds (Wiens and Rotenberry 1981), or in cases where vegetation structure and composition are highly correlated.

Distinct patterns in the relationship between bird abundance and structural features related to rangeland health in my study echo those previously established in the literature (Bock *et al.* 1993; Madden *et al.* 2000; Fisher and Davis 2010) and outline the range of habitat conditions required by the grassland bird community. Baird's, grasshopper and savannah sparrows and Sprague's pipit, whose abundances generally increased with vegetation cover, are typical of light to moderately grazed dry mixed grasslands (Madden *et al.* 2000; Wheelwright and Rising 2008). Horned lark, chestnut-collared longspur, and McCown's longspur, whose abundance decreased with greater amounts of vegetative cover and increased with bare ground cover, are typically associated with moderate to heavy grazing (With 1994; Beason 1995; Davis *et al.* 1999). This partitioning of the grassland bird community along a continuum of habitat features may be attributed to their evolution with grazing by large herbivores on the Northern Great Plains (Knopf 1996) and the subsequent niches resulting from the heterogeneity in vegetation structure, competitors, and prey (Wiens 1973). I did not find strong relationships between structural variables related to rangeland health and abundance for vesper sparrow or western meadowlark. Both species occupy a wide range of grassland types (Jones and Cornely 2002; Davis and Lanyon 2008) and the variation of vegetation structure represented in my study likely fell within their niche requirements.

Although vegetation structure was an important predictor of abundance in my study, my results suggest that plant species richness and within-habitat heterogeneity are also important for some species. Plant species richness and within-habitat heterogeneity are not considered important drivers of grassland bird abundance, nor are they frequently examined in field studies (Fisher and Davis 2010). My results are consistent with others in that grasshopper sparrow was associated with somewhat homogenous litter cover and vegetation height (Wiens 1973) and patchy bare ground (Vickery 1996) whereas horned lark was associated with homogenous bare ground cover (Beason 1995). Plant species richness may represent greater structural complexity (Fisher and Davis 2010) or food availability (Wiens 1969; Rodenhouse

1981; Wiens and Rotenberry 1981; Sample 1989) while structural heterogeneity may provide a range of available shelter, nesting or foraging sites (Wiens 1974 a, b; Fuhlendorf *et al.* 2006).

2.5 Implications

I demonstrate that while rangeland health itself is not a strong predictor of bird abundance, structural components used to assign this index are. If the rangeland health index is to be used to assess bird habitat, I recommend that it be altered to reflect the importance of vegetation structure. For example, greater value could be attributed to refined categories of litter, and vegetation volume (Robel *et al.* 1970) could be added to the rangeland health index. I advocate for future research that examines relationships between rangeland health and grassland biodiversity and determines the extent to which indices of rangeland health are currently used by rangeland managers on private and public lands and whether using such measures would be feasible for private livestock producers.

PREFACE TO CHAPTER 3: VOLUNTARY STEWARDSHIP AND THE CANADIAN SPECIES AT RISK ACT: EXPLORING RANCHER WILLINGNESS TO SUPPORT SPECIES AT RISK IN THE CANADIAN PRAIRIES

North American temperate grasslands and wildlife species they support are increasingly imperilled, largely due to habitat loss and degradation. Most remaining prairie is privately managed and supports livestock production. In Canada, voluntary stewardship is the preferred approach for protecting species at risk on private lands under the federal Species at Risk Act (SARA). However, attitudes of private land managers towards species at risk and their willingness to engage in stewardship are poorly understood. Therefore, the second objective of this thesis was to describe producer characteristics, summarize producer awareness of and attitudes towards species at risk and the Canadian Species at Risk Act and evaluate how characteristics, awareness and attitudes affect producer willingness to engage in voluntary stewardship actions that support species at risk conservation.

I used a mixed-methods approach and data from interviews with 42 livestock producers in Saskatchewan, Canada, to describe producer characteristics, attitudes and awareness of species at risk and evaluate how these factors influence willingness to protect species at risk. Younger producers with increased formal education, awareness and positive attitudes were more willing to support conservation of species at risk. Voluntary stewardship under the SARA may be enhanced by rewarding producers for sound habitat management and improving trust between producers and government agencies.

Chapter 3 is published in the journal Human Dimensions of Wildlife.
See: Henderson, A.E., M. Reed and S. K. Davis. 2014. Voluntary Stewardship and the Canadian Species at Risk Act: Exploring Rancher Willingness to Support Species at Risk in the Canadian Prairies. *Human Dimensions of Wildlife: An International Journal*.19:17-32.

CHAPTER 3: VOLUNTARY STEWARDSHIP AND THE CANADIAN SPECIES AT RISK ACT: EXPLORING RANCHER WILLINGNESS TO SUPPORT SPECIES AT RISK IN THE CANADIAN PRAIRIES

3.1 Introduction

In North America, temperate grasslands and the wildlife species they support are increasingly imperilled, with an estimated 57 prairie species considered at risk of extinction (IUCN 2011). In Canada, 28 prairie species are federally listed as species at risk (Committee on the Status of Endangered Wildlife in Canada (COSEWIC) 2012) and 30 prairie species are federally listed as endangered in the United States (United States Fish and Wildlife Service 2012). These declines are largely due to the loss and degradation of native prairie habitat (Samson and Knopf 1996; Venter *et al.* 2006). Approximately 21% of native prairie remains in North America, much of it lost to cultivation for agricultural production, urbanization, and extensive energy sector development (White *et al.* 2000). The majority of extant North American prairie provides important habitat for species at risk and exists on privately managed lands (Ramunkuty *et al.* 2008; White *et al.* 2000 ;). In the United States, over 90% of the species protected under the federal Endangered Species Act (1973) (ESA) have habitat on non-federal lands (General Accounting Office 1994). Similarly, in Canada, private agricultural lands provide important habitat for species at risk (Kerr and Deguise 2004).

Livestock production is the predominant land-use on remaining native prairie (Tanaka *et al.* 2005b) and can enhance or degrade habitat via changes to the structure and function of rangeland plant communities (Fuhlendorf and Engle 2001). Cattle grazing can create a mosaic of grass species and vegetation structure that provides habitat for a wide variety of wildlife species (Derner *et al.* 2009). Alternatively, it can create irreversible changes to rangeland and riparian plant communities, impact ecosystem functioning and degrade habitat (Freilich *et al.* 2003). Given the predominance of livestock production and the capacity of cattle grazing to shape grassland habitat, livestock producers and the grazing management decisions they make play a key role in species recovery on private lands.

Although the US and Canada share similar prairie species and habitats, their legislative approach to species protection on private lands differs (Illical and Harrison 2007; Olive 2012). In Canada, the federal Species at Risk Act (2005)(SARA) supports voluntary stewardship as the preferred approach for conserving species at risk on private lands and is in line with the use of voluntary policy instruments in federal resource management in other sectors (Winfield 2009). Voluntary stewardship includes any voluntary action taken by a person to protect species at risk. There are several mechanisms to protect species, their residences and critical habitat under the SARA (Mooers *et al.* 2010). Once listed, all individuals and their residences on federal lands receive protection under the SARA's general prohibitions. On private lands, however, only individuals and residences of listed migratory birds or aquatic species are protected. Once a species' critical habitat has been identified by the federal government, the habitat receives immediate protection from destruction on federal lands or if it is considered aquatic. If provincial or territorial laws do not effectively protect the species' critical habitat or no such protection is in place, the federal government may apply the SARA's general prohibitions through an emergency order after consultation with affected stakeholders. In the case of an emergency order or where critical habitat has been identified on private lands, the Minister may provide compensation to private land managers for losses suffered. At the time of preparing this thesis, only one emergency order has been registered to come into effect Feb. 18, 2014 (for Greater Sage Grouse) (Her Majesty the Queen in Right of Canada 2013), however compensation for losses suffered on private lands where critical habitat has been identified has not yet been applied (Wojciechowski *et al.* 2011). Voluntary stewardship efforts for listed species on private lands are supported by the federal Habitat Stewardship program, which provides funding and facilitates partnerships for activities that protect and conserve species at risk and their habitat.

On private lands in the United States, conservation under the ESA generally places more emphasis on enforcing regulations rather than voluntary stewardship. Costs of species protection are largely borne by private firms or individuals rather than the government (Illical and Harrison 2007; Raymond and Olive 2008). Efforts to make the ESA more incentive-based and less coercive for private individuals materialized in the 1990s through amendments such as "Safe Harbour" agreements and the "No Surprises" rule (Goble *et al.* 2006).

Few studies have examined the protection of species at risk on private lands in Canada (e.g., Olive 2012; Wojciechowski *et al.* 2011). While studies have investigated the perspectives of private landowners on endangered species recovery under the ESA model in the United States (Reading and Kellert 1993; Benson *et al.* 1999; Willcox and Giuliano 2011), none has examined the factors underlying willingness of private landholders to participate in voluntary stewardship in Canada. For species at risk protection on private lands to be effective under the SARA, it is important to learn how Canadian producers perceive species at risk and what drives producers' willingness to protect species at risk on their lands.

3.1.1 Analytical Framework

Studies from different disciplines have examined factors underlying participation of agricultural producers or private landholders in voluntary stewardship (e.g., Decker *et al.* 2001; Chouinard *et al.* 2008; Didier and Brunson 2004). Personal characteristics, awareness and knowledge, and attitudes emerge as key factors that affect the voluntary participation of producers (Kabii and Horwitz 2006; Pannell *et al.* 2006).

In the present study, I examined how age, education, duration of land tenure and land-holding size affect producer voluntary participation. I considered the influence of producer awareness and knowledge on willingness to protect species at risk because inadequate knowledge or awareness of a policy or innovation is known to limit its uptake (Kabii and Horwitz 2006). Finally, I explored producer attitudes towards species at risk and species at risk policy because such attitudes are good predictors of behaviour (Ajzen and Fishbein 2005). Using a mixed methods approach, my study addressed knowledge gaps by examining factors underlying livestock producer willingness to protect prairie species at risk in southwestern Saskatchewan, Canada. My objectives were to (a) describe producer characteristics, (b) summarize producer awareness of and attitudes towards species at risk and the SARA and (c) evaluate how characteristics, awareness and attitudes affect producer willingness to engage in voluntary stewardship actions that support species at risk conservation.

3.2 Methods

I used a mixed methods approach that combines quantitative and qualitative research to broaden my understanding by converging broad, numeric trends with detailed, contextual information (Creswell 2009).

3.2.1 *Study region*

I selected the Milk River watershed of southwestern Saskatchewan, Canada as my study region as it contains the highest diversity of species at risk in the province and the largest tracts of privately managed native rangeland (Figure 1-1). Native rangeland in the study region is either under private, provincial or federal management which comprise 70%, 20% and 10% of the area, respectively (Henderson unpublished data). In this study, private land managers were chosen as interview participants by randomly selecting quarter sections (i.e., 160 acres or 65 ha) of native rangeland based on the Dominion Land Survey system (McKercher and Wolfe 1986). I used ArcGIS 9.3 (Environmental Systems Research Institute 2008) to stratify my random sampling based on private, provincial and federal land-management and selected 140 quarter sections of upland native prairie with loam or solonetzic soils. Federally, provincially and privately managed lands were evenly distributed across the study region (Figure 1-2). I only used Rural Municipality maps to identify names of private land managers, herein referred to as producers, for each randomly selected quarter section and contacted them by phone. In total, 42 producers agreed to be interviewed (representing 94% of all producers contacted).

3.2.2 *Sampling Design and Data Collection*

Questions addressed the personal ranching history of producers, awareness, knowledge and attitudes towards species at risk and the SARA and their willingness to support species at risk recovery through management actions. All interviews were conducted with the primary land-manager(s) of the household in winter months of 2009 and 2010. Interviews were digitally recorded and ranged from 30 minutes to 85 minutes (average 50 minutes). Recordings were then transcribed *verbatim* with Express Scribe version 5 (NCH Software 2012) by three trained assistants and checked for accuracy. Interview participants are represented herein by a randomly assigned code (i.e., P1 through P100) to maintain their anonymity.

3.2.3 *Mixed-Methods Approach*

I analysed all data using NVivo (Version 9) (NVivo 2010).I quantified responses to categorical interview questions to summarize producer characteristics and awareness, and used matrix coding queries and cluster analysis to evaluate willingness to engage in species at risk recovery actions. My qualitative approach was inductive, which allowed my research findings to emerge from dominant themes. I assigned attribute categories (i.e., producer age, duration in region, duration ranching, number of generations ranching, and size of land holdings) to each interview and summarized producer characteristics with coding matrix queries. To describe producer awareness, knowledge and attitudes, I assigned words, phrases, sentences or stories within the interview dialogue to emerging thematic codes, also referred to as "nodes" in NVivo (Creswell 1998). I classified attitudes as positive, conditional or negative based on the level of support for species at risk and the SARA expressed in producer responses. I determined overall willingness to support species at risk recovery by examining willingness to (a) share information about species at risk on their lands, (b) adopt a new management strategy, and (c) participate in species at risk recovery planning. I evaluated how characteristics, attitudes and awareness relate to indicators of willingness using coding matrix queries and cluster analyses of node coding similarity to give greater depth to the qualitative analysis.

Within NVivo*,* cluster analysis involved construction of a matrix where columns were interviews and rows were selected nodes (i.e., either characteristics, awareness, or attitudes and willingness). Cells of the matrix consisted of 0 or 1; a value of 1 was assigned if the interview was coded by the selected node and a value of 0 was assigned if not. Then, a similarity index between each pair of nodes in the matrix was calculated using the Pearson correlation coefficient. Nodes were grouped into clusters using a complete linkage hierarchical clustering algorithm which computed clusters based on the maximum distance between node similarity indices. We named clusters according to emerging themes, determined the number of interview responses coded to each node using the group query tool and examined characteristics of producers represented in each cluster (i.e., age, land holding size) using the attribute classification sheet and group query results in NVivo.

3.3 Results

3.3.1 *Personal Characteristics*

My analysis revealed minor variation in producer characteristics. Ninety percent of producers who self-identified as the primary land-manager in the household were male; the remaining 10% were husband and wife teams who both identified as land-managers. Fifty-five percent were 3rd generation producers, 93% had at least one previous generation of their family involved in ranching (Figure B1-A). Sixty four percent of producers were 50-59 years old (Figure B2-B) and had lived in the region for more than 50 years (Figure B1-C). Fifty two percent of producers had ranched for 30-39 years (Figure B1-D). Eighty eight percent of producers in our study managed at least 1536 ha of rangeland; 62% managed more than 2590 ha (Figure B1-E).

Sixty-nine percent of producers interviewed learned to manage their land by experience or through the experience of family members previously involved in ranching. Thirty-one percent of producers learned to manage their land through both education and experience. The most important sources of information for making management decisions were talking with family and neighbours (identified by 69% of producers) and general experience (identified by 67% of producers). The “Internet, newspapers or magazines” and “workshops or education” were identified as the next most important sources of information by 48% and 40% of producers, respectively.

3.3.2 *Awareness of Species at Risk and SARA Policy*

All producers had heard of species at risk and 95% were aware of the SARA. Most were unfamiliar with details of the SARA or the species listing process; 38% did not know where the SARA applies and 50% thought that the SARA applies on all lands (i.e., federal, provincial and private). Only 7% knew that general prohibitions of the SARA apply to all species and their residences on federal lands. Sixty-four percent of producers replied that species at risk are identified by a count process of some kind, yet only 10% identified COSEWIC as the committee responsible for recommending species listings under the SARA. Thirty-eight percent had previously participated in a wildlife stewardship program and 33% had used a land-management strategy to support a species at risk.

Despite low levels of awareness regarding species at risk policy and process, many producers were familiar with prairie species at risk and their associated recovery efforts and expressed detailed knowledge of species at risk that live on their lands. When asked to name three prairie species at risk, 90% could name at least two. When asked "What kinds of wildlife use your lands?", 38% of producers described details of habitat use, population changes or behavioural observations of wildlife on their lands.

3.3.3 *Attitudes towards Species at Risk and the SARA*

Seventy-nine percent of producers conveyed positive attitudes towards species at risk and the SARA and showed support for government involvement in species at risk recovery. Thirty-three percent said that regional prairie species listed under the SARA should be considered at risk due to declining populations; many conveyed personal observations of declines in particular wildlife species. Seventy-six percent of producers placed conditions on their support of species at risk, replying that declining species should be considered at risk as long as there were no changes to producers' economic well-being or management approach. Twelve percent of producers supported government involvement in species at risk recovery on the condition that government work *with* producers using an approach that is "not heavy-handed". A few were doubtful that many prairie species at risk should be considered at risk (4%), either because some species are at the northern-most extent of their range or the producer had not observed declines in the listed species. A small subset of producers expressed negative attitudes towards the listing of wildlife species as at risk (14%). They did not support government involvement in species at risk recovery (2%) and did not support the listing of prairie wildlife as species at risk (14%) either because they thought the species was not originally found in the region (e.g., swift fox), they observed fluctuations in the population over time that they consider normal, or they felt the *species at risk* label could draw unnecessary and potentially harmful attention to a species:

> "I think that having some of them, like the burrowing owls and that [as listed species], does more harm for them than if they were left alone. As soon as you

get into this risk thing then you got people coming, wanting to see 'em….where if they weren't being advertised as at risk, they'd be just another bird." (P17)

Four predominant attitude themes emerged from my dialogue with producers about species at risk and the SARA, which I've classified as "producers as stewards", "economic risk", "lack of trust and communication", and "do not disturb" (Table 3-1).

Eighty-six percent of producers believed that prairie wildlife species, including those considered at risk would not be in the region if it were not for the stewardship efforts of producers over the last 100 years (Table 3-1). Several interviewees suggested that livestock producers could be rewarded by the government for the habitat they provide species at risk through their management and maintenance of native prairie (Table 3-1). In this way, species at risk could be considered assets for ranching operations rather than liabilities.

Sixty-six percent of producers identified their ranching operation as a business and would support species at risk recovery through a stewardship program or new management strategy only under conditions that did not impart increased economic risk (Table 3-1). Sixty percent of producers specifically identified money or capital as the primary constraint to adopting new management techniques. Several expressed the difficulty of making a living as a producer and identified pride for their lifestyle or a "love of the land" as the only reasons to continue ranching. At the time interviews were conducted, producers felt cattle prices needed to improve if they were going to abate economic risk and do more than "squeeze their lands for every last nickel". Fifty-seven percent of producers identified economic inputs as the most important incentive for adopting new management strategies to support species at risk recovery.

Many producers expressed fear, lack of trust and poor communication with government on issues surrounding species at risk (55%) (Table 3-1). Producers expressed a widespread fear of the "strong arm of the law" that could be applied through the SARA to increase regulations on their lands, thereby compromising their independence as managers. Forty percent of producers expressed the need for improved communication with government on issues of species at risk, explaining that the atmosphere surrounding government-producer communication is "more like us against them right now" (Table 3-1). Producers felt that the government's role in recovery planning on private lands was to work *with* producers, rather than forcing regulations or telling producers how to manage.

Twenty-one percent of producers expressed fear that sharing information about species at risk on their lands would lead to “more encroachment of unwanted people”; they said they would like to be left alone to manage their lands the way they choose (Table 3-1). Thirty-six percent also explained that species at risk should be left alone and expressed concern over how increased attention from researchers or nature enthusiasts might affect species’ well-being. Maintaining privacy on their lands and minimizing harm to species were the primary conditions producers placed on sharing information about species at risk on their lands after “economic risk”.

3.3.4 *Making Connections with Willingness*

Overall, producers were willing to support species at risk recovery, often under the conditions that producer privacy, independence, and the financial stability of their operation be maintained and wildlife species not be harmed by increased attention. When I considered all indicators of willingness (i.e., to share information about species at risk on their lands, adopt a new management strategy, and participate in regional species at risk recovery planning), matrix coding queries identified producers aged 60-69 (Figure B2-A) and those who had ranched for more than 50 years (Figure B2-B) as less willing to support species at risk than producers in other categories. Matrix coding queries for indicators of willingness and producer approach to learning about rangeland management, awareness, and attitudes revealed patterns that warranted further exploration with cluster analyses. Two groups emerged from my cluster analysis of willingness and producer approach to learning about rangeland management (Figure 3-2A). Producers who learned via some level of formal education (31%) expressed willingness to support species at risk recovery more often than producers who learned to manage their lands based on experience alone (69%). A greater proportion of producers in the “experience” group were 60-69 years old, had spent at least 30 years in the region, and managed larger land holdings (i.e., 83% managed more than 2590 ha) than producers in the “education” group. By contrast, a greater proportion of “education” producers were 40-49 years old, had spent less than 20 years in the region and managed smaller land holdings (i.e., 77% managed < 2590 ha).

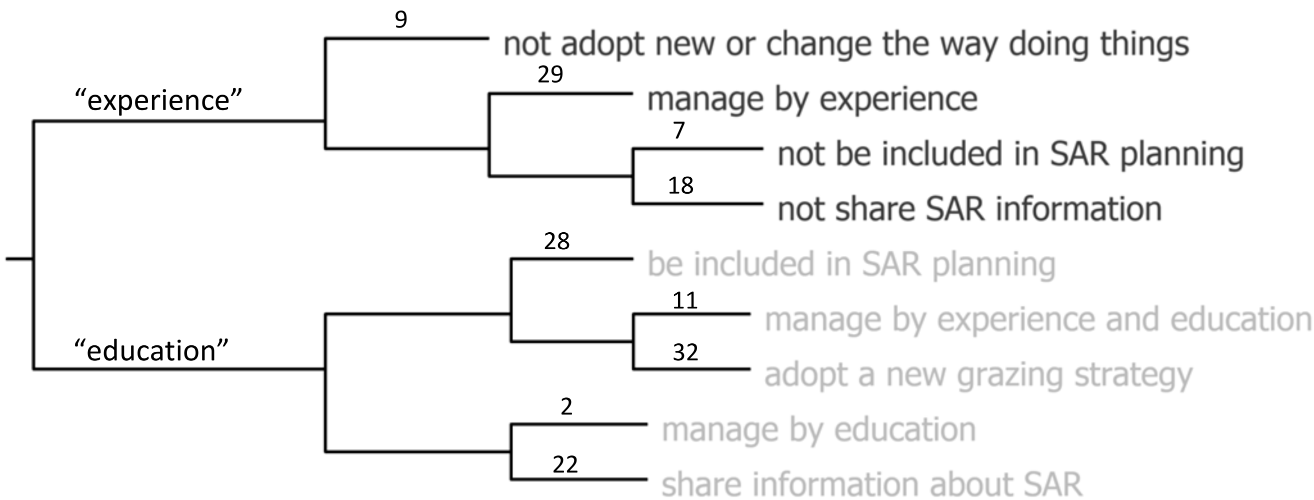
Two distinct groups emerged in my cluster analysis of willingness and awareness of species at risk and the SARA (Figure 3-2B). Producers who responded that "the SARA applies on all lands" expressed less willingness to share information about species at risk on their lands, adopt new management strategies or be included in species at risk recovery planning, while interviewees who revealed "greater detailed knowledge" were more often coded with greater overall willingness. Producers who expressed specific knowledge of species at risk on their lands were more willing to share information of species at risk on their lands, despite not knowing precisely where the SARA applies. Those not aware of where the SARA applies more frequently expressed a willingness to be included in species at risk recovery planning or adopt a new grazing management strategy. Producers with specific knowledge of the species at risk listing process were also more familiar with where the SARA applies. The two awareness clusters that emerged could not be differentiated on the basis of producer characteristics, as responses from individual producers were distributed across both groups.

Two groups, classified as "positive or conditional" and "negative", emerged from my cluster analysis of attitudes towards species at risk and willingness (Figure 3-2C). Producers with positive and conditional attitudes towards species at risk showed greater willingness to share information about species at risk on their lands, adopt new grazing strategies and be included in species at risk recovery planning than producers who expressed negative attitudes. "Positive or conditional" and "negative" producers differed in the time spent ranching in the region and in the size of the landholdings they manage. Eighty-three percent of producers in the "negative" group spent more than 50 years in the region and all ranched for more than 30 years. By contrast, 42% of "positive or conditional" producers lived in the region for less than 50 years and 24% had ranched for less than 30 years. Eighty-three percent of producers in the "negative" group managed more than 2590 ha of land while 45% of producers in the "positive or conditional" group managed less than 2590 ha.

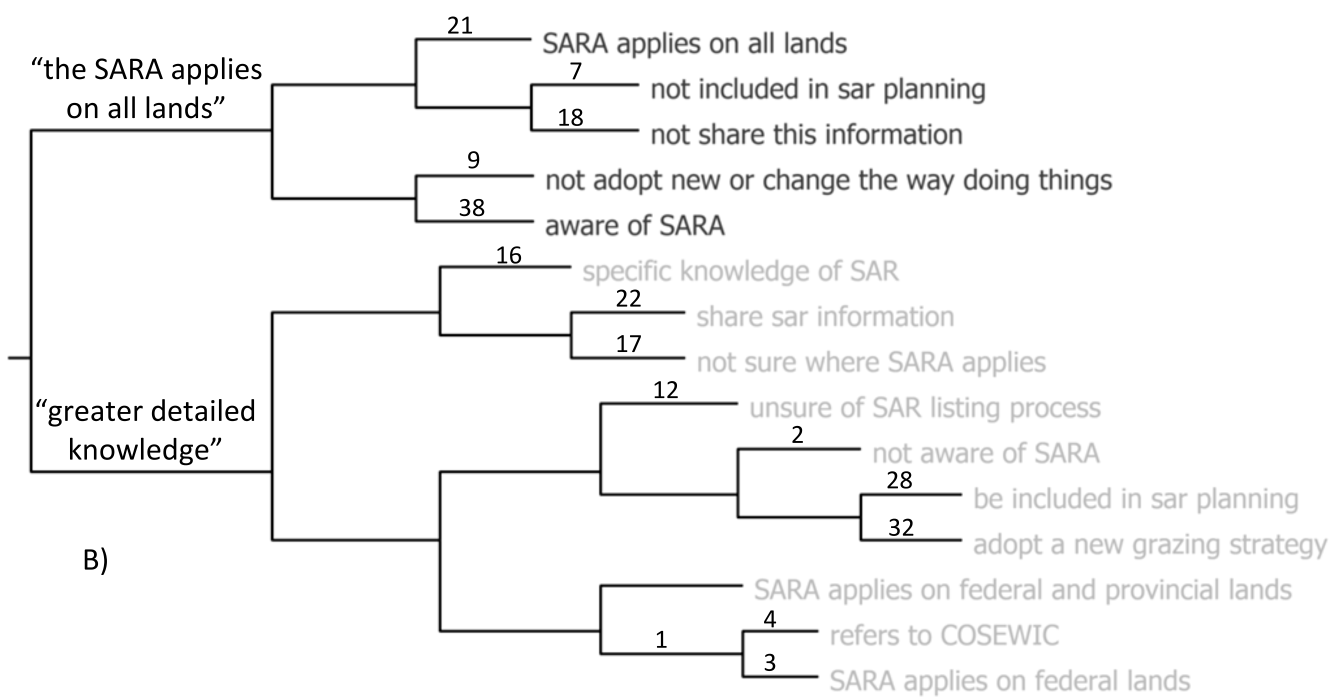
Table 3-1. Supporting quotations for five key attitude themes identified from qualitative analysis of interview responses: “producers as stewards”, “economic risk”, “lack of trust and communication”, “do not disturb” and “rewarding ranchers”.

Theme	Supporting Quotations
Producers as Stewards	“Well the native prairie…we're just part of it. We believe that the native prairie should stay there. And we do watch it, I don't know how you would say…guard it, we are aware of it.“ P19 “It's pretty obvious that the ranching community has done a pretty good job of looking after the wildlife for the last 150 years. Or there wouldn't be any of it left here anymore.” P58
Economic Risk	“I've ranched for 30 years for no profit to support 9000 acres of species at risk...and if somebody can show me a better way to do that and I would do it, and I'd also appreciate if they'd show me how to earn a living while doing that.” P28 “The understanding needs to be there that we are trying to run a business here, so any rules or regulations affect how we run our business.” P13 “This has to be financially feasible for us. Because we can't sacrifice our life and our livelihood and everything we've worked for…you're fighting for fifty years to keep things paid for. You're at the whim of nature and the world market. It's not easy, you know. It looks easy, but it isn't. “ P36
Lack of Trust and Communication	“Well, you know what I'm thinking is as soon as they report species at risk, you know, their antenna goes up and pretty soon the government's gonna wanna take away their lease land. Jeez. You've got so much government involvement now I don't wanna see anymore.” P50 “I would be very careful how I shared it, I'll tell ya…because of some of the ramifications that could happen, I suppose…If you had some over-zealous person, you know...say there happened to be an owl in the middle of a hay field or something, all of a sudden there's $20,000.00 of hay there and all of a sudden they said whoa, stop, you're not gonna cut that this year. Well, then…that's the worst thing that could happen…and that causes people to keep their mouth shut.” P36
Do not Disturb	“If I did [share information about species at risk] I'd just get a flock of people bothering me here. Just leave me alone.” P50 “Yeah, well if you tell people there are species at risk you get a bunch of people coming around and they figure they're helping but actually they're disturbing the poor things…like the burrowing owls I didn't [share that information] because one other person had a burrowing owl, and then all of a sudden there was a whole bunch of people coming around and the little owl took off cause he didn't like all this attention.” P10 “I would [share information about species at risk] as long as there's not too much action that happens out there. I don't mind helping anybody, but not if they're gonna make a big deal and bring a whole pile of people in and scare the little beggars away.“ P39
Rewarding Ranchers	“I think a Species at Risk should be considered an asset…something that is worth your while having. So, if I had them, and if they are an asset then theoretically I should be rewarded for maintaining and enhancing that asset…rather than being subjected to rules and regulations.“ P2 “If ranchers were compensated on the basis of the environmental goods and services they provided, it would actually improve the habitat for all wildlife and would put money in the pockets of the rancher. I want to see something that will recognize good stewardship on the land that has already taken place, and to use the classifications that are in place [to assess the health of the rangeland].” P2

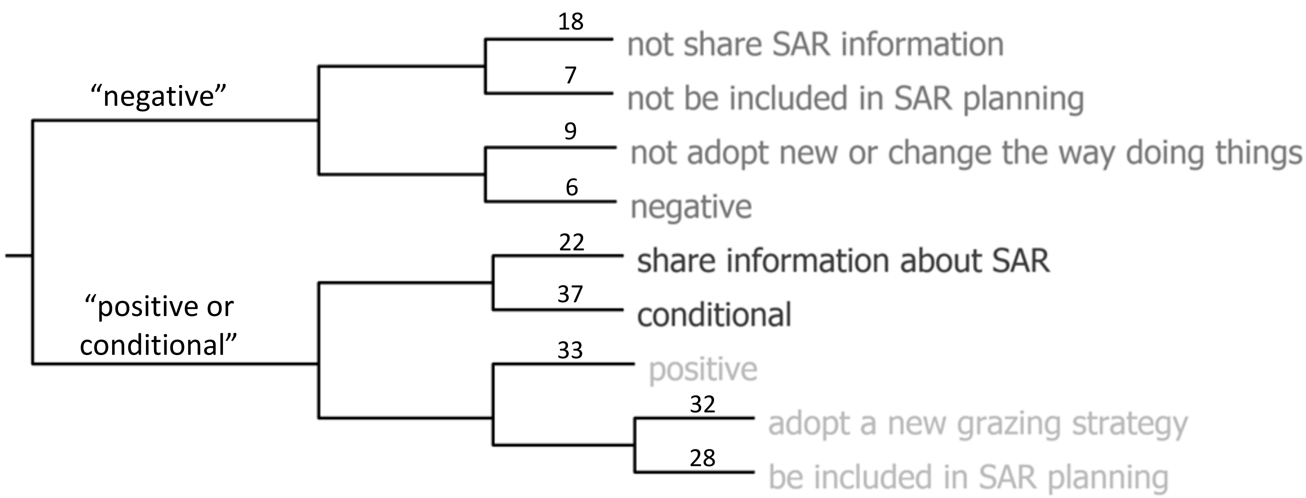
Figure 3-2. Horizontal dendrograms displaying results of cluster analyses of indicators of willingness and A) how producers learned to manage their rangeland, B) producer awareness of species at risk and the Species at Risk Act, and C) attitudes towards species at risk. All cluster analyses are based on node coding similarity using the Pearson correlation coefficient similarity index. Nodes that appear close together are more similar than those that are far apart. Numbers that appear on branches indicate the number of producers whose responses were coded for that particular node. Note that ranchers may be coded to more than one node.



A)



B)



C)

3.4 Discussion

I identified age, education, awareness and knowledge, and attitudes towards species at risk as important to producer willingness to participate in voluntary stewardship actions under the SARA. My results agreed with previous studies that examined factors underlying voluntary stewardship of agricultural producers on private lands (e.g., Kabii and Horwitz 2006). Younger ranchers with some level of formal education, greater awareness and knowledge of species at risk policy were more willing to support species at risk recovery.

Most land managers of ranching families in the study region were males who learned to manage their lands by experience. Ranching is a largely patrilineal and highly adaptive enterprise (Bennett 1969; Starrs 2002); livestock producers typically acquire rangeland management knowledge by experience working alongside their fathers and older ranchers (Bennett 1969; Starrs 2002; Knapp and Fernandez-Gimenez 2009). Similar to Knapp and Fernandez-Gimenez (2009), I found that, along with experience, talking with family and neighbours was an important information source for making rangeland management decisions. Social networks appear to be well-established in my study region, as in other ranching communities (Decker 2001; Mathijs 2003). Land-holdings were slightly larger in my study region compared to the provincial average, likely due to the fact that ranches in southwestern Saskatchewan require a large land base to graze cattle amidst frequent drought (Sauchyn *et al.*

2010). The majority of ranchers in my study were nearing retirement (i.e., aged 50-69), which is consistent with national trends for agricultural producers in general (Statistics Canada 2011).

Similar to case studies under the ESA model in the US (Raymond and Olive 2008), I encountered a low level of producer awareness regarding details of species at risk legislation. Very few producers were clear on how species at risk are listed or what the implications of the SARA are on privately managed lands. They did, however, express detailed knowledge of their lands as habitat for wildlife and were familiar with local population changes and behaviours of species at risk on their lands. According to Knapp and Fernandez-Gimenez (2009) this local, “embedded knowledge” gained by ranchers living on the land and observing natural processes can complement scientific knowledge in the development of management practices.

The largely positive attitudes producers held towards species at risk reinforces a strong stewardship ethic that producers expressed in my interviews. Ranchers may hold positive attitudes towards wildlife’s intrinsic value and express concern for threatened or endangered species (Conover 1998; Kellert 1980; Willcox and Giuliano 2011) or they may express doubt in whether threatened and endangered species should be listed at all. As I observed, negative attitudes towards species at risk may be aimed more frequently towards the public agencies that implement endangered species policy, rather than towards species themselves (Clark *et al.* 1994). As in my study, ranchers may place conditions on their support of conservation, avoiding actions that compromise their economic welfare or impose further government regulations (Kabii and Horwitz 2006; Conley *et al.* 2007). The four themes that emerged in my qualitative analysis are consistent with themes arising in the literature. As in previous works (Kellert 1980; Sayre 2002; Willcox and Giuliano 2011), ranchers in my study considered stewardship of native prairie and its wildlife as implicit to ranching. Many credited previous generations of ranchers with maintaining native prairie habitat for wildlife. Ranching was portrayed as more than merely a business, rather, it was described as a way of life that includes a connection to the land (Decker 2001; Sayre 2002). Lifestyle attributes linked with stewardship, such as family tradition, culture and values, are often ranked above profit maximization as motives for ranching (Gentner and Tanaka 2002).

My findings also agree with previous studies that identify economics as an important, but not always primary, motive for adopting management innovations to support wildlife stewardship (Tanaka *et al.* 2005a). Economic constraints of agricultural operations can enable or limit producers' willingness to participate in conservation measures (Conley *et al.* 2007). Given the high input costs and low rates of return on investment in recent years, profit is hardly a top-ranking motive for ranching (Tanaka *et al.* 2005a). As such, some consider cattle ranching as a financially stressful and economically irrational enterprise (Bennet 1969; Sheridan 2007).

Agricultural producers and private landholders commonly expressed fear of increased regulations and lack of trust or communication with the public agencies responsible for implementing stewardship programs for endangered species (Brooke *et al.* 2003; Conley *et al.* 2007). This tension between ranchers and the government on issues surrounding endangered species is long-standing in the United States (Clark *et al.* 1994), and arises from a combination of differences in rancher and agency views, fear of government intrusion on private property rights, and inadequate collaboration with ranchers on issues surrounding endangered species (Decker 2001; Conley *et al.* 2007; Jackson-Smith *et al.* 2005). Establishing respect and trust between producers and public agencies is an important component of improving producer participation and resolving the social dilemmas of rangeland management (Mathijs 2003; Van Kooten *et al.* 2006). To surpass the "us against them" atmosphere, public agencies can invite producers to participate in program planning, increase face-to-face time and informally engage as individuals, rather than as representatives of organizations (Wondolleck and Yaffee 2000).

Ideas captured in my "do not disturb" theme also echo those in the literature. Ranchers value privacy and choose to live in places where there are few people; solitude and the independence to make management decisions without government intervention are key elements of the ranching lifestyle (Decker 2001; Starrs 2002). Agricultural producers who perceive endangered species legislation and other forms of government intervention as threats to their lifestyle and private property rights are less likely to participate (Conley *et al.* 2007). The concern expressed by my producers that species at risk could be disturbed by increased attention reveals "embedded" local knowledge of how species many respond to research activities (Knapp and Fernandez-Gimenez 2009) and a clear lack of communication between wildlife biologists and ranchers regarding the rationale and animal care procedures of wildlife research.

Of the personal characteristics I examined, producer age, length of time spent ranching and level of education demonstrated the strongest influence on willingness to support species at risk. My finding that producers aged 60-69 were less willing to support species at risk agrees with previous work that identifies older producers as more risk averse, unwilling to change their management approach and less aware of management innovations (Potter and Lobley 1992). Furthermore, landowners aged 60-69 are nearing retirement and have much of their resources tied up in their land; this may also make them less willing to adopt changes that may compromise their ability to sell their land or transfer it to an heir. Producers who learned to manage their lands with some level of formal education were generally more willing to support species at risk. As Kabii and Horwitz (2006) postulate, this may be because they are more informed. Awareness is a key influence of producer willingness to engage in voluntary stewardship (Kabii and Horwitz 2006; Atari *et al.* 2009). Greater awareness of an issue may allow producers to determine whether engaging in a program or practice will align with their management goals, which, according to Pannell *et al.* (2006), is imperative to landholder adoption of a conservation practices. My study supports previous research showing that positive attitudes increase producer willingness to engage in voluntary stewardship because they play a key role in predicting behaviour (Ajzen and Fishbein 2000; Willcox *et al.* 2012).

3.5 Implications

My results identify ways to facilitate mechanisms for protecting species at risk on private ranch lands under the SARA. Since younger producers with more formal education and positive attitudes towards species at risk may be more willing to engage in voluntary stewardship actions, stewardship action plans and conservation agreements could focus outreach on younger ranchers (i.e., < 60 yrs old) and/or establish a specific strategy for engaging older ranchers (i.e., > 60 yrs old) who are less willing to support species at risk. While a high level of stewardship for species at risk already exists in the study region, rancher compliance with general prohibitions of the SARA may be enhanced by improving levels of awareness of species at risk policy. Ranchers need outreach that communicates how government agencies are involved in the implementation of the SARA, how species at risk are listed, what the species at risk recovery process involves and how the SARA might affect private landowners. This could be achieved using the structure of social networks in the ranching community, as this is known to influence the uptake of voluntary stewardship programs (Rogers 1995; Didier and Brunson 2004). The agencies responsible for implementing the SARA could improve conservation successes by applying the local knowledge of ranchers to the development of management strategies and best management practices (Knapp and Fernandez-Gimenez 2009; Willcox *et al.* 2012).

My findings suggest that conserving species at risk on private ranch lands through voluntary stewardship under the SARA is feasible, given certain conditions. Many producers in my study region identify themselves as stewards of their land and are willing to support species at risk conservation but expressed concern for the financial implications. Producer willingness to support species at risk conservation could be encouraged by improving levels of trust between producers and public agencies responsible for implementing the SARA. Species at risk programs could achieve success by recognizing rancher stewardship for native prairie and rewarding ranchers with economic support or incentives for the ecosystem goods and services they provide. Programs that rely on payments for ecosystem services are a useful tool for achieving biodiversity protection goals, provided that their design considers the appropriate type and scale of the market institution, captures multiple ecosystem services, and eliminates the potential of perverse effects caused by strategic behaviours that undermine the intended conservation goal (Banjeree *et al*., 2013; Kronenberg and Hubacek, 2013).

PREFACE TO CHAPTER 4: MODELING SOCIAL AND ECOLOGICAL DRIVERS OF ABUNDANCE FOR THREE GRASSLAND SONGBIRDS AT RISK

Temperate native grasslands are among the most altered and imperilled ecosystems on the planet and provide important habitat for rare species. Effective protection of grassland fauna requires an understanding of how both social and ecological factors pertaining to rangeland management affect grassland habitat and wildlife abundance. Hence, the final objective of this thesis was to elucidate novel pathways for achieving grassland bird conservation by exploring relationships between select social and ecological factors and abundance for three species considered at risk of extinction: Chestnut-collared longspur (*Calcarius ornatus*), McCown's longspur (*Rhynchophanes mccownii*), and Sprague's pipit (*Anthus spragueii*).

I used an interdisciplinary approach and structural equation model to explore how social and ecological variables relate to habitat and abundance for the three aforementioned grassland bird species in southwestern Saskatchewan, Canada. I based my model on data from avian point counts, vegetation surveys, rangeland health assessments and interviews with managers on federal, provincial and privately managed grasslands. My three focal bird species did not respond uniformly to vegetation structure or social variables. Sprague's pipit abundance was better supported by lands managed federally than those under provincial or private management. Additionally, Sprague's pipit abundance was greatest in habitat managed by managers of large land holdings (> 2590 ha) and with increasing litter (kg · ha -1), and vegetation volume (cm^3). Habitat with decreasing litter and vegetation volume had the highest abundance of Chestnut-collared and McCown's longspurs. McCown's longspur abundance was also greater in habitat characterised by managers who held negative attitudes towards species at risk. Therefore, I conclude that conservation of the grassland bird community may be best supported by a patchwork of public and private management which, through variation in grazing pressure, could create a mosaic of vegetation structure and habitat conditions. I also demonstrate how social and ecological data can be integrated to inform conservation and management of imperilled species using structural equation modeling.

Chapter 4 is in preparation for submission.

See: Henderson, A. E., E. Lamb, S. K. Davis, and M. Reed. 2014. Modeling Social and Ecological Drivers of Abundance for Three Grassland Songbirds at Risk. In preparation for submission to *Conservation Biology*.

CHAPTER 4: MODELING SOCIAL AND ECOLOGICAL DRIVERS OF ABUNDANCE FOR THREE GRASSLAND SONGBIRDS AT RISK

4.1 Introduction

Habitat loss and degradation are primary causes of species endangerment that have contributed to the decline of approximately 85% of endangered wildlife species worldwide (Baillie *et al.* 2004). Human land-use, including agricultural production, is the leading cause of habitat loss in North America (Wilcove *et al.* 1998; Czech *et al.* 2000) and temperate grasslands are among the most severely affected ecosystems. Conversion to agricultural production has converted or degraded an estimated 41% of the Earth's temperate grasslands, causing dramatic declines in grassland species (White *et al.* 2000). Approximately 57 grassland wildlife species are currently considered at risk in North America, 28 of which are grassland birds (IUCN 2011).

Livestock production, the predominant land-use on remaining native prairie (Tanaka *et al.* 2005b; Ramankuty *et al.* 2008), can enhance or degrade habitat and rangeland health[4] via changes to the structure and function of rangeland plant communities (Fuhlendorf and Engle 2001). Cattle grazing can create a mosaic of plant species and vegetation structure that provides habitat for a wide variety of grassland species (Michulnas *et al.* 1998; Vavra 2005). Alternatively, it can create potentially irreversible changes to grassland and riparian plant communities, negatively impact ecosystem function and degrade wildlife habitat (Freilich *et al.* 2003). Given the predominance of livestock production in grassland regions and the capacity of grazing to shape the health of grasslands, livestock producers and the management decisions they make are instrumental to maintaining and improving habitat for imperilled grassland species (Maestas *et al.* 2003).

Social factors are significant determinants of conservation outcomes and necessary components of conservation science (Mascia *et al.* 2003; Balmford and Cowling 2006; Fox *et al.* 2006). Understanding human actions towards the environment requires an understanding of

[4] Rangeland health indices are a standard tool for assessing grassland structure and community composition and indicate how close producers are to achieving optimal grassland health on a particular ecological site defined by soil and site stability, hydrologic function, and biotic integrity (Pyke *et al*. 2002; Adams *et al*. 2005; Pellant *et al*. 2005).

human values, attitudes and knowledge (Clark *et al.* 1994; Manfredo 2008). Values and attitudes can reliably predict human behaviour (Kraus 1995; Ajzen and Fishbein 2005) and inadequate knowledge or awareness of a policy or innovation is known to limit its uptake (Kabii and Horwitz 2006). On private rangelands that support livestock production, conservation outcomes may rely on the voluntary stewardship[5] of producers, which can be influenced by personal characteristics, such as producer age, size of land holdings, knowledge of the conservation issue, financial security or lifestyle, and social factors, such as social networks and societal norms or attitudes (Didier and Brunson 2004; Kabii and Horwitz 2006; Moon *et al.* 2012). A clear understanding of these factors enables voluntary stewardship programs to be tailored to increase producer participation and habitat protection (Kabii and Horwitz 2006; Henderson *et al.* 2014). Effective habitat protection, therefore, requires a clear understanding of how both social and ecological factors influence habitat and biodiversity (Mascia *et al.* 2003; Forester and Machlis 2006).

Multivariate relationships between social and ecological variables can be difficult to assess using traditional approaches (Meffe and Viederman 1995; Forester and Machlis 2006; Grace 2006). Structural equation models provide a useful method of integrating social and ecological data and understanding relationships between them (Forester and Machlis 2006; Grace 2006; Brewer *et al.* 2012). Structural equation models are a versatile statistical modeling tool based on general linear modeling procedures that can be used to evaluate multivariate hypotheses (Grace 2006). Researchers have used structural equation modeling to examine either social or ecological dimensions of conservation problems, including geographical gradients of species extinction risk (Lee and Jetz 2011), species-habitat relationships (Iriodono *et al.*2003; Baldwin *et al.* 2007; Ficetola *et al.* 2011) and social drivers of participation in conservation (Austin *et al.* 1998; Willock *et al.* 1999). However, few researchers have examined relationships between social and ecological drivers of biodiversity loss (Mora 2008).

My objective was to explore relationships between social factors (i.e., management jurisdiction, manager age, attitude towards species at risk, willingness to support species at risk recovery and size of land holdings), ecological factors (i.e., vegetation structure and rangeland

[5] Voluntary stewardship includes any voluntary action taken by a person to protect species or habitat.

health), and grassland songbird abundance. In doing so I aimed to elucidate novel strategies to manage native grassland habitat for the conservation of grassland species, in this case grassland birds. I used structural equation modelling to explore relationships between social and ecological factors that I postulated were important drivers of conservation and habitat selection for three grassland bird species considered at risk of extinction in Canada: Chestnut-collared longspur (*Calcarius ornatus*), McCown's longspur (*Rhynchophanes mccownii*), and Sprague's pipit (*Anthus spragueii*).

4.2 Methods

4.2.1 Data collection

4.2.1.1 Study Area

I selected the Milk River watershed in southwestern Saskatchewan, Canada as my study area as it contains the highest diversity of species at risk in the province and the largest tracts of native rangeland grazed by livestock (Figure 1-2). I stratified my random sampling of 140 quarter sections (i.e., 65 ha each) of upland native prairie based on land-cover and soils data (Beckingham *et al.* 1996) and on the amount of native grassland under private (70%), provincial (20%) and federal (10%) management in the region (see Henderson and Davis 2014 for details). However, because I did not interview all managers of the selected quarter sections, my sampling scheme for the current study represented 64% private, 11% provincial and 25% federally managed lands. My sample quarter sections represented rangelands managed by private ranchers (78 quarter sections), three provincial community pastures (14 quarter sections), six federal community pastures (21 quarter sections) and one national park (10 quarter sections). Provincial community pastures were managed by the Saskatchewan Pastures Program of the Saskatchewan Ministry of Agriculture. Federal community pastures were managed by the Agri-Environment Services Branch of Agriculture and Agri-Food Canada. Native grassland in the national park was managed by staff of Grasslands National Park. Private lands were managed by individual ranchers, ranching families or ranching cooperatives. Upland native mixed-grass prairie in the study region was characterized by either loam or solonetzic soils and mixed-grass or fescue prairie communities, largely dominated by *Elymus lanceolatus* (Scribn. and Sm.) Gould, *Pascopyrum smithii* (Rydb.) Barkworth and D.R. Dewey, *Calamagrostis montanensis* (Scribn.) Vasey, *Festuca hallii* (Vasey) Piper, *Festuca saximontana* Rydb., *Hesperostipa comata ssp. comata* (Trin. and Rupr.) Barkworth, or *Hesperostipa curtiseta* (Hitchc.) Barkworth (Thorpe 2007; ITIS 2013).

4.2.1.2 Avian Surveys

I randomly positioned centres of three 100 m radius point-count circles 300 m apart and at least 100 m from habitat edges within each quarter section (Hutto *et al.* 1986). Point count surveys were conducted from 26 May to 3 July in 2009 and 2010 from 0.5 hrs before sunrise until four hours after sunrise on mornings with wind <20 kph and no precipitation. Trained surveyors conducted one 5-minute count at each survey location by recording aurally and visually detected singing males within 100m from the observer. I estimated distance to each bird (Buckland *et al.* 2001) and recorded bird detections within three equal time periods during the survey period (Farnsworth *et al.* 2002). I used the sum of male birds aurally detected within 100 m over all three point counts as an index of abundance. Analysis of both distance and removal sampling data did not warrant adjustments to estimates of abundance (Henderson and Davis 2014). I used only those individuals detected by song to estimate the number of territorial males breeding in each quarter section because I could not reliably separate females from non-singing males for most species. While I recorded all detected species, I only considered abundances of three species considered at risk of extinction in my study area: Chestnut-collared longspur, McCown's longspur, and Sprague's pipit.

4.2.1.3 Vegetation Measurements

In each quarter section, I measured plant species composition, erosion, litter and vegetation volume in 24 plots (25 x 50 cm frame) distributed regularly along the lines between the three bird point count locations (Figure A-1). In each plot, I estimated the percentage that each plant species contributed to total plant biomass within the plot; I used these data to calculate species richness. I measured vegetation volume using a Robel pole with 2.5 cm increments (Toledo *et al.* 2008) and estimated 100% obscurity to the nearest cm in all cardinal directions. All measurements were assessed from 4 m away at a height of 1 m (Robel *et al.* 1970). I visually estimated litter mass (kg·ha-1) by hand raking to collect all dead plant material (e.g., standing stems, fallen stems, and leaf material and partially decomposed material) within the plot and compared this to a litter normal typical of the range site being evaluated (Adams *et al.* 2005; Saskatchewan PCAP Greencover Committee 2008). I visually estimated signs of erosion and percent cover of club moss (*Selaginella* spp.), lichen, and bare ground (i.e., any land surface not covered by vegetation). I measured shrub cover (%) using the line-intercept method (Canfield 1941) on three 100 m transects randomly located between point count centers. I used measurements of plant species composition, erosion and litter to assign each quarter section an index of rangeland health using the Saskatchewan Rangeland Health Index (Saskatchewan PCAP Greencover Committee 2008). The final Rangeland Health score represented rangeland that can be broadly classified as “unhealthy” (<50), “healthy with problems” (50-75), or “healthy” (75-100); for full details see Henderson and Davis (2014).

4.2.1.4 Interviews

I used Rural Municipality maps to identify land managers for each randomly selected quarter section and contacted them by phone. In total, 55 land managers representing 123 quarter sections agreed to be interviewed in the winter months of 2009 and 2010. Face-to-face Interviews were digitally recorded, transcribed *verbatim* with Express Scribe (Version 5) (NCH software 2012) and checked for accuracy. My semi-structured interviews were analyzed by quantifying categorical responses related to manager characteristics and awareness of species at risk, and conducting qualitative analysis of open-ended interview questions related to manager knowledge, attitudes and willingness to support species at risk.

4.2.2 Data Analysis

Structural equation modeling enables researchers to conceptualize and evaluate complex relationships between multiple intercorrelated variables (Grace 2006). First, a diagram of the initial model that outlines the hypothesized relationships among the categorical or continuous variables is translated into linear equations. When the model includes only observed or measured variables, it is called an observed variable structural equation model. The independent and dependent variables are referred to as exogenous and endogenous variables, respectively (Grace 2006). Whereas exogenous variables only influence endogenous variables, endogenous variables can, in turn, influence other endogenous variables. Endogenous categorical variables can pose important issues due to the increased potential for non-normality of residuals, however exogenous categorical variables pose no real problems because the distributional assumption of normality of the residuals does not apply (Grace 2006). My social variables were exogenous and categorical; all other variables were endogenous and continuous (Table 4-1).

Table 4-1. Descriptions and summary statistics of the categorical and continuous variables included in the structural equation model; ··· indicates that the calculation is not applicable.

Variable	Description	Range	Frequency	Percentage	Mean	Standard Deviation
FED	federal management (i.e., Agriculture and Agri-Food Canada or Parks Canada); categorical	0 (non-federal) 1 (federal)	0: 85 1: 31	0: 73 1: 27	···	···
PROV	provincial management (i.e., Government of Saskatchewan Community Pasture); categorical	0 (non-provincial)[6] 1 (provincial)	0: 103 1: 13	0: 89 1: 11	···	···
ATT	manager attitudes towards species at risk and the Species at Risk Act; categorical	0 (negative) 1 (positive)	0: 17 1: 99	0: 15 1: 85	···	···
WILL	producer willingness to support species at risk; categorical	-3 to 3	-3: 6 -1: 6 0: 5 1: 51 3: 48	-3: 5 -1: 5 0: 4.5 1: 44 3: 41.5	···	···
AGE	producer age; categorical	29 to 69 years	35: 12 45: 30 55: 64 65: 10	35: 10 45: 26 55: 55 65: 9	···	···
SIZE	size of land holdings (ha); categorical	1 (1 to 1295 ha) 2 (1295 to 2590 ha) 3 (>2590 ha)	1: 3 2: 16 3: 97	1: 2.5 2: 14 3: 83.5	···	···

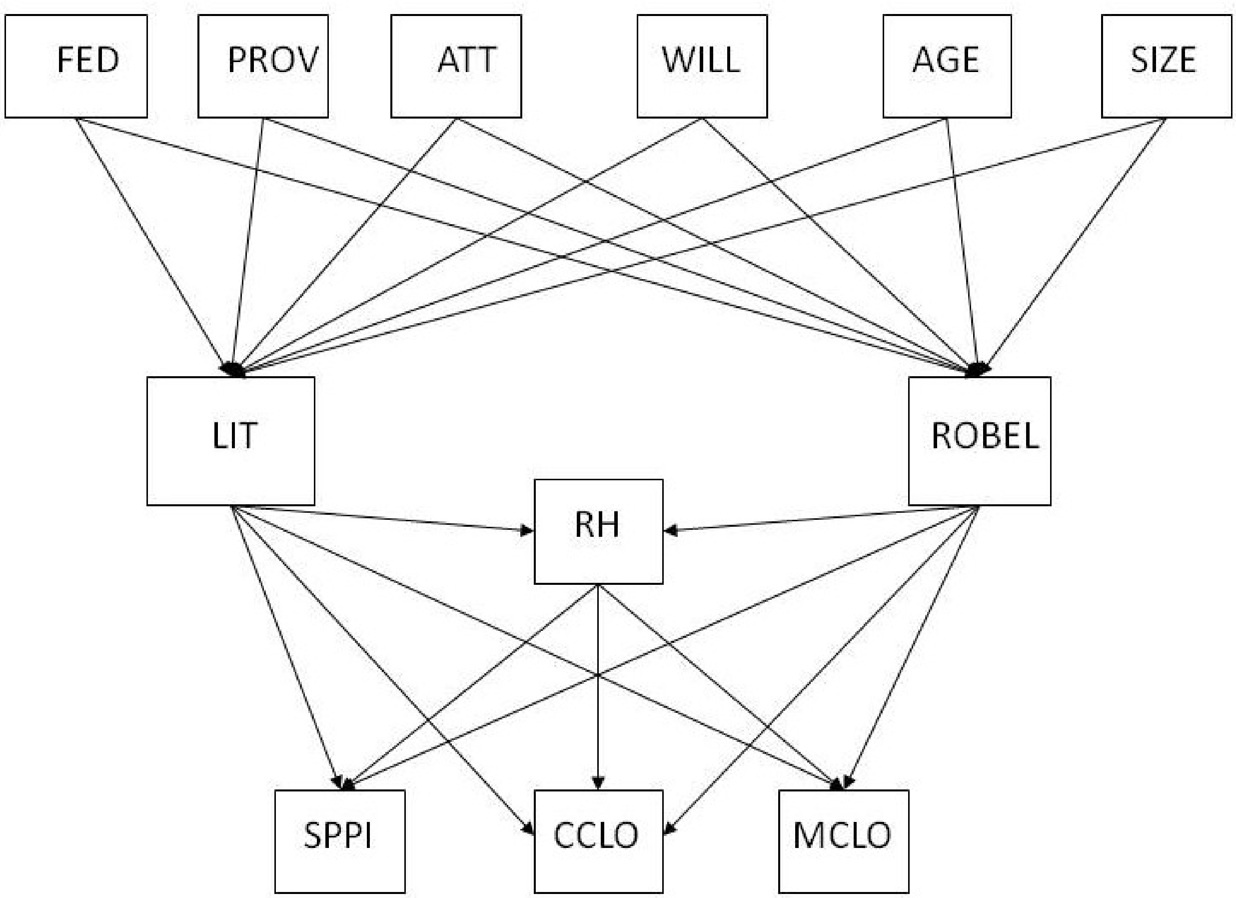
[6] Privately managed quarter sections provided a baseline for assigning management jurisdiction and received a "0" for both FED and PROV categories. All relationships from FED and PROV categories are compared to privately managed lands.

LIT	litter mass (kg·ha-1); continuous	7 to 1000	···	···	192.76	209.03
ROBEL	vegetation volume (cm^3); continuous	6 to 62	···	···	19.19	8.43
RH	rangeland health score (/100); continuous	24 to 90	···	···	61.02	16.13
CCLO	Chestnut-collared longspur abundance; continuous	0 to 17	···	···	5.60	4.08
MCLO	McCown’s longspur abundance; continuous	0 to 13	···	···	1.27	2.16
SPPI	Sprague’s pipit abundance; continuous	0 to 10	···	···	2.01	2.45

4.2.2.1 Model specification

I developed an initial model comprised of observed social (Henderson *et al.* 2014) and ecological (Henderson and Davis 2014) variables that I found to be the most influential to grassland bird abundance based on previous studies (Figure 4-1). I included social variables such as management jurisdiction (i.e., federal, provincial or private), manager age, size of land holdings (ha), attitudes towards species at risk and willingness to support species at risk recovery (Table 4-1). I hypothesize that these variables affect grassland bird habitat and abundance through their influence on rangeland management. I included litter and vegetation volume on the basis of my previous work that identified litter and vegetation volume as important predictors of bird abundance (Henderson and Davis 2014). I included the rangeland health index because it captures a wider array of management outcomes (e.g. hydrologic function and site stability, plant community composition) than vegetation structure alone. I included direct paths from all social variables to litter, vegetation volume and the rangeland health index to account for the effects of social variables on rangeland management. I included direct paths from litter, vegetation volume and rangeland health to abundance for my three focal bird species to account for the fact that bird abundance is influenced by vegetation structure (Knopf 1996; Fisher and Davis 2010).

Figure 4-1. Initial structural equation model of relationships between manager characteristics, habitat features and Sprague's pipit (SPPI), Chestnut-collared longspur (CCLO), and McCown's longspur (MCLO) abundance. Social factors include federal (FED) and provincial (PROV) management jurisdictions, manager attitude towards species at risk (ATT), manager willingness to support species at risk recovery (WILL), manager age (AGE) and size of land holdings under management (SIZE). Ecological variables include litter amount (kg·ha-1) (LIT), vegetation volume (cm^3) (ROBEL) and rangeland health score (RH). The direction of the arrow indicates the direction of the relationship between two variables. For example, federal management (FED) has a direct effect on litter (LIT) and vegetation volume (ROBEL).



4.2.2.2 Model fitting

I examined the bivariate correlations between all variables and checked for linearity in these relationships using the software R (Version 2.15.2) (The R Statistical Computing Group 2012). I fit all models using M-Plus (Version 7) (Muthen and Muthen 2012). I assessed model fit by comparing the expected covariance structures derived from the initial model to covariance structures from a variance-covariance matrix of the variables in the dataset (Lamb *et al.* 2011). Since each land manager could manage more than one sample unit (i.e., quarter section), I assigned land manager as a cluster variable using the 'TYPE=COMPLEX' option in M-Plus to account for non-independence in the data. I assessed model fit with a χ^2 test and used modification indices to identify novel paths that could improve model fit. Modification indices are an estimate of the change in χ^2 expected if a new path is added to the model (Lamb *et al.* 2011). In using modification indices, my structural equation model became exploratory rather than confirmatory (Grace 2006).

4.2.2.3 Model interpretation

Path coefficients represent partial regression coefficients when two variables are connected by more than one pathway. I calculated the total net effect when two variables were connected via both indirect and direct paths. The total effect of one variable on another was the sum of path coefficients from both direct and indirect effects (Grace 2006). Indirect effects from compound paths that included multiple links (e.g., FED→RH→SPPI, Figure 4-2) were calculated as the mathematical product of path coefficients from each path. I considered standardized path coefficients statistically significant at $p < 0.05$ and relationships with path coefficients 0-0.3 as weak, 0.4-0.5 as moderate, and 0.6-0.8 as strong.

4.3 Results

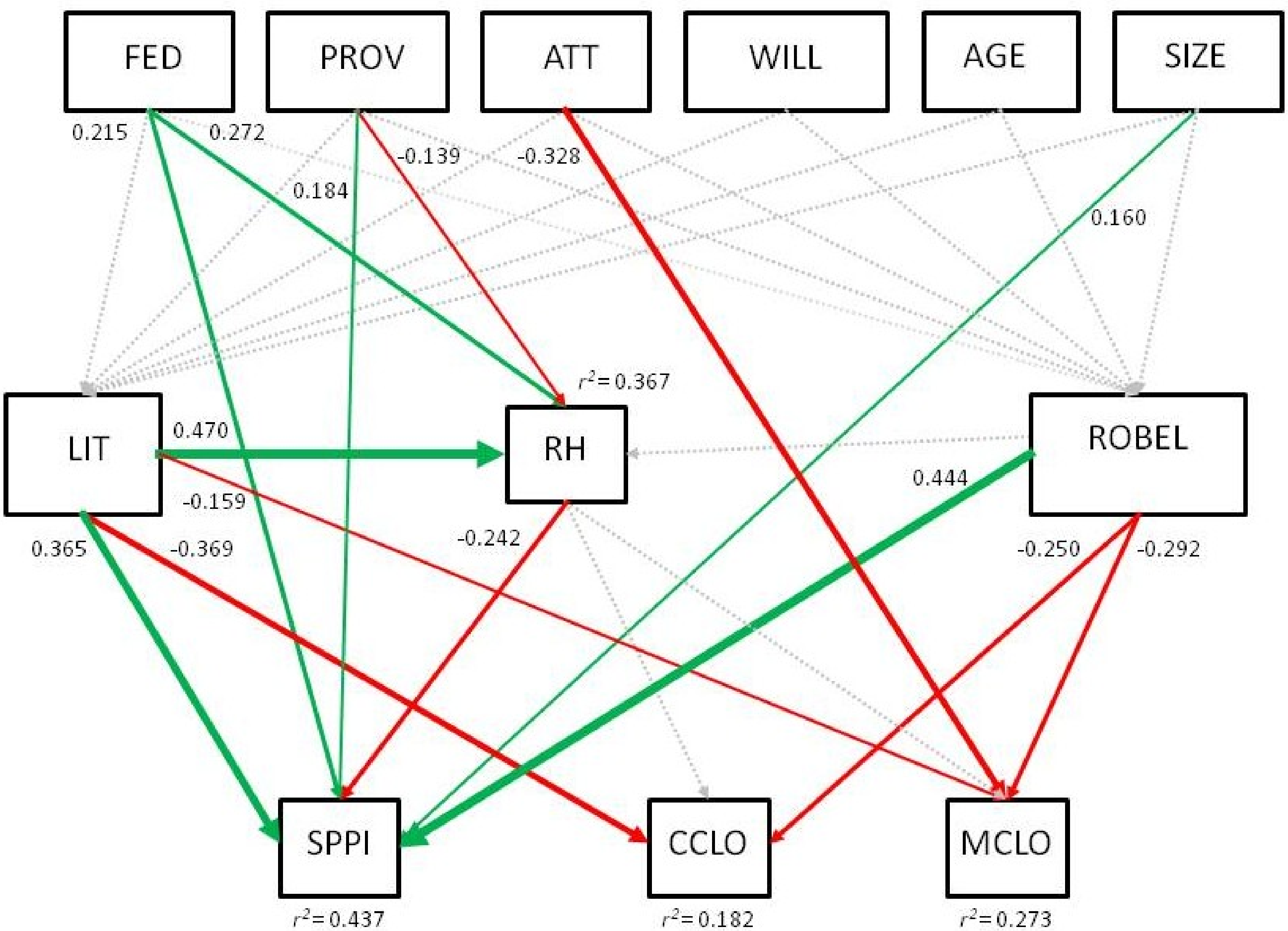
4.3.1 Model fit

The initial model of bird abundance (Figure 4-1) did not demonstrate adequate fit (χ^2= 83.15, p<0.01). Modification indices suggested additional direct paths from management jurisdiction to rangeland health, from attitude to McCown's longspur abundance and from management jurisdiction and size to Sprague's pipit abundance (Figure 4-2) (Grace 2006). Since federal, provincial and private managers have different mandates and resources for management, including paths from management jurisdiction to rangeland health and to Sprague's pipit abundance seemed logical. The justification for the modified path from attitude to McCown's longspur abundance was less clear. However, the direction of this effect was in line with my previous findings that indicate McCown's longspur prefer habitat characteristic of relatively heavy grazing pressure (Henderson and Davis 2014). If negative attitudes towards species at risk correlate with a management approach that emphasizes economic gain rather than habitat protection, then it is plausible that negative attitudes towards species at risk could result in habitat suitable for McCown's longspur. Since Sprague's pipit abundance is known to increase with increasing patch size (Davis 2004), the modified path from size to Sprague's pipit abundance was justified. The fit of the modified model was adequate once the additional paths were included (χ^2= 25.68, p=0.14).

4.3.2 Social variables

My modified structural equation model identified management jurisdiction (i.e., federal, provincial or private) as important to rangeland health and to Sprague's pipit abundance (Figure 4-2). Compared to privately managed lands (x=57, s=14, n=57) rangeland health was greater on federal lands (x=68, s=11, n=8) and lower on provincial lands (x=49, s=10, n=3) (Table 4-2, Figures 4-2, 4-3). On federal lands, 8, 45, and 47% of quarter sections were classified as "unhealthy", "healthy with problems", or "healthy", respectively. On provincial lands, 64, 32, and 5% of quarter sections were classified as "unhealthy", "healthy with problems", or "healthy", respectively. On private lands, 42, 43, and 15% of quarter sections were classified as "unhealthy", "healthy with problems", or "healthy", respectively. Sprague's pipit abundance was greater on federal and provincial lands compared to those under private management (Figures 4-2, 4-3) and was also greater on federal compared to provincial lands (Figure 4-3). The total net effects of federal and provincial management on Sprague's pipit abundance were 0.149 (i.e., 0.215 + (0.272 * -0.242) = 0.149) and 0.218, respectively (Figure 4-2). Sprague's pipit abundance also increased with increasing land holding size (Table 4-2, Figure 4-2), which varied with management jurisdiction (Figure 4-3). Land holding size was larger on federal and provincially managed lands, compared to private lands (Figure 4-3). McCown's longspur abundance increased with increasingly negative manager attitudes towards species at risk (Figure 4-3). However, manager attitude was not correlated with litter, vegetation volume or rangeland health, indicating that the mechanism by which attitude affects McCown's longspur abundance (e.g., bare ground cover, shrub cover) was not captured in my model.

Figure 4-2. Final structural equation model of relationships between manager characteristics, habitat features and Sprague's pipit (SPPI), Chestnut-collared longspur (CCLO), and McCown's longspur (MCLO) abundance. Social factors include federal (FED) and provincial (PROV) management jurisdictions, manager attitude towards species at risk (ATT), manager willingness to support species at risk recovery (WILL), manager age (AGE) and size of land holdings under management (SIZE). Ecological variables include litter amount (kg·ha-1) (LIT), vegetation volume (cm^3) (ROBEL) and rangeland health score (RH). The direction of the arrow indicates the direction of the relationship between two variables. Statistically significant paths representing a positive effect appear in green and those representing a negative effect appear in red. The thickness of the solid arrows reflects the magnitude of the standardized path coefficients which are listed beside each significant path. Dashed grey lines indicate non-significant paths.



4.3.3 Ecological variables

Sprague's pipit abundance increased with increasing vegetation volume and litter (Table 4-2) (Figure 4-2). The total net effect of litter on Sprague's pipit abundance was 0.251 (0.365 + (0.47*-0.242) = 0.251) and the direct effect of vegetation volume on Sprague's pipit abundance was 0.444 (Figure 4-2). Chestnut-collared and McCown's longspur abundance decreased with increasing litter and vegetation volume (Table 4-2) (Figures 4-2, 4-4). Greater litter mass was associated with habitat higher in rangeland health. Rangeland health was not a significant driver of abundance for either longspur species. Rangeland health was a statistically significant driver of Sprague's pipit abundance, however its effect was weak and in a reverse direction to the bivariate relationship (Figures 4-2, 4-4). Overall, the weak to moderate r^2 values we report for all endogenous variables indicate that the explanatory variables we selected do not explain all of the variation in our data (Figure 4-2).

Figure 4-3. Statistically significant relationships between variables in the modified final model (p <0.05) including relationships between McCown's longspur abundance and attitude, management jurisdiction, rangeland health score, Sprague's pipit abundance and size of land holdings (ha); FED= federal, PRIV= private and PROV= provincial management. In each box-and-whisker plot, the top and bottom of the box represent the first and third quartile of the data, respectively, the band inside the box represents the median, the upper and lower dashed "whiskers" represent the maximum and minimum data values, respectively, and dots represent outliers.

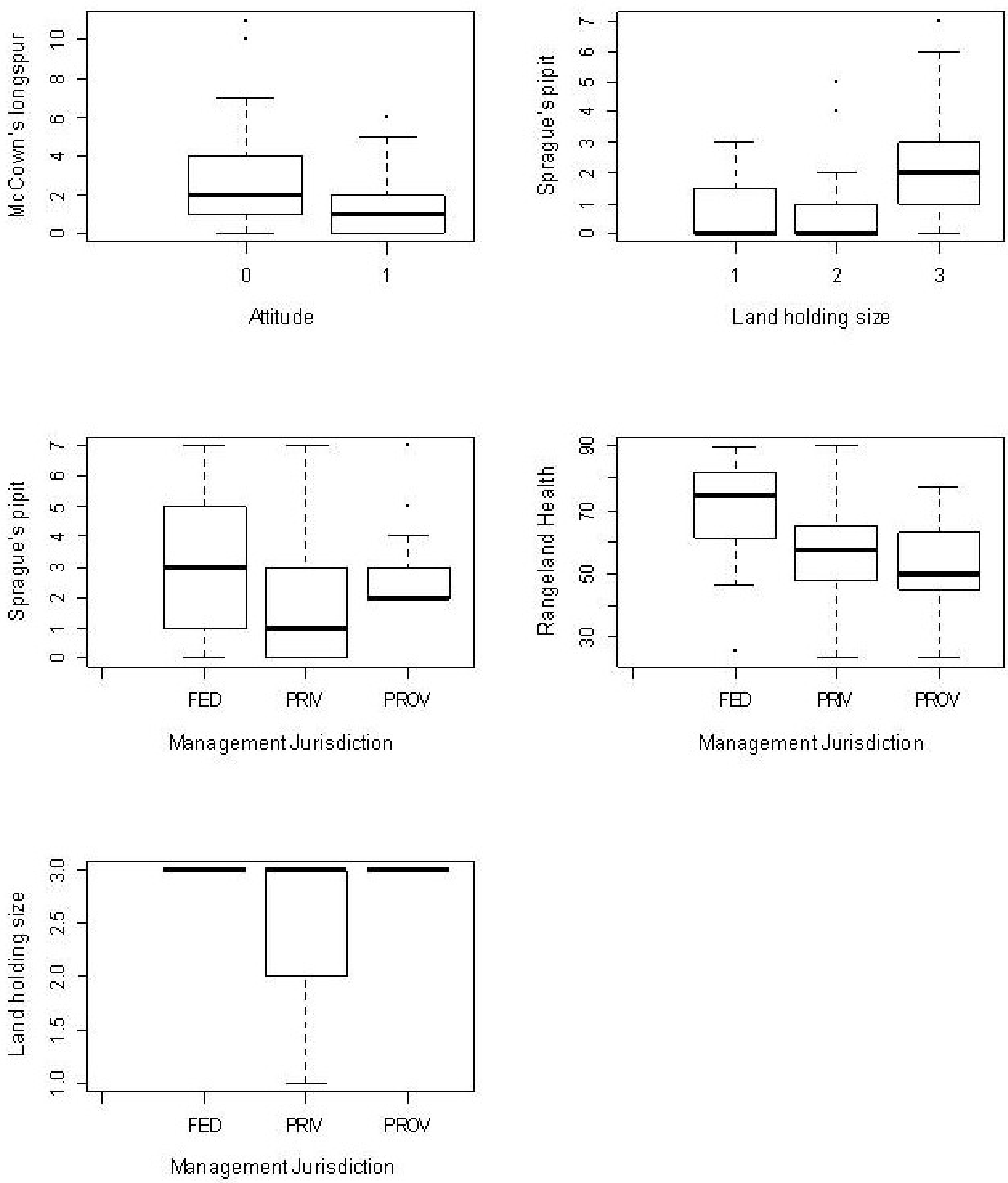


Figure 4-4. Scatterplots representing statistically significant relationships between variables in the modified final model (p<0.05). Grey lines represent linear regressions.

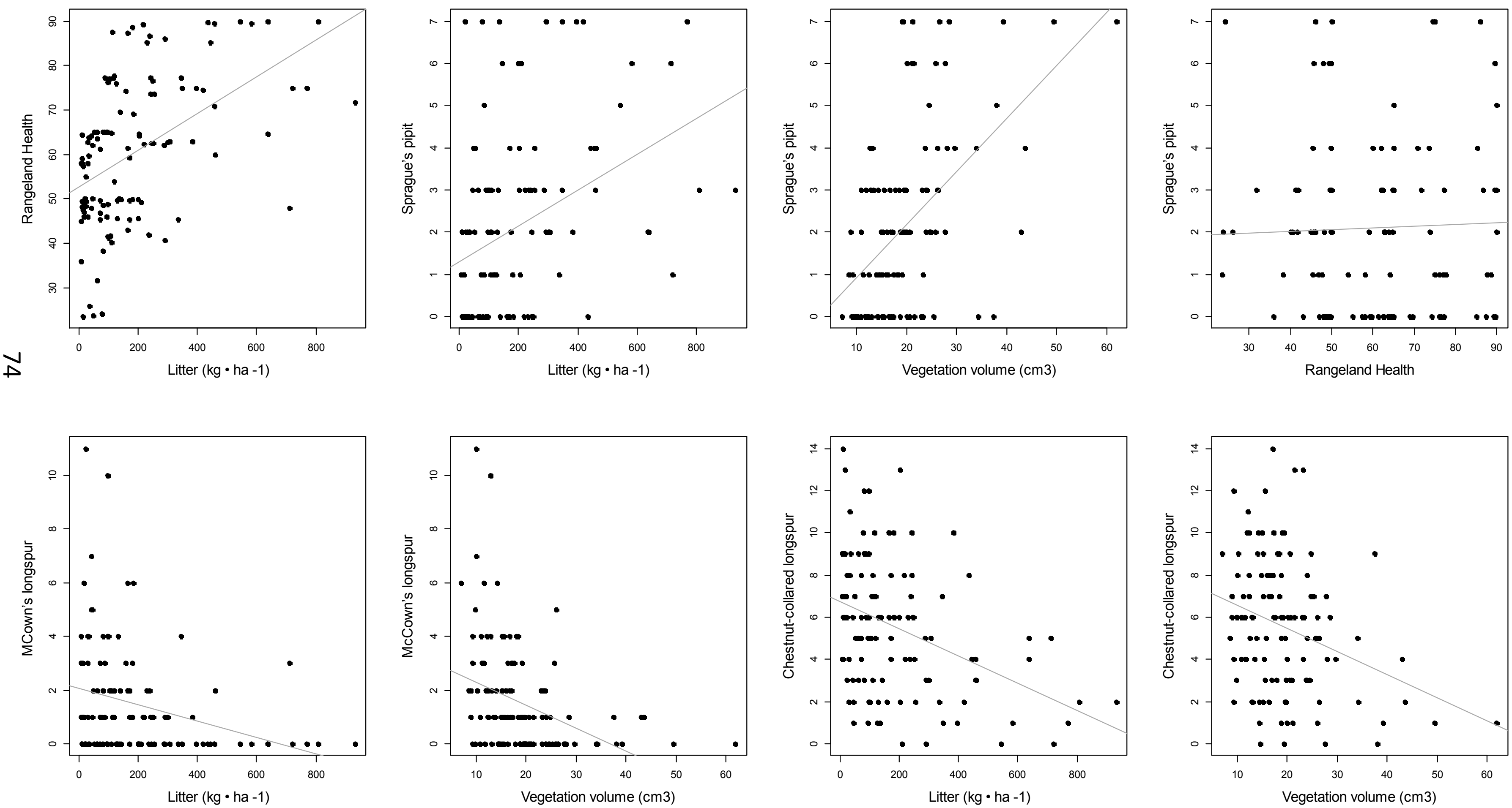


Table 4-2. Unstandardized and standardized path coefficients, the standard error of the unstandardized coefficients and Z test results from the observed path model in Figure 4-2. Social variables include management jurisdiction, manager attitude towards species at risk, manager willingness to support species at risk recovery, manager age, size of land holdings under management. Ecological variables include litter amount (kg·ha-1), vegetation volume (cm^3) and rangeland health score. Paths in the model were from the non-bold variables to the variable in bold at the top of each section in the table. Standardized estimates are in standard deviation units and are used to compare the relative strengths of paths. Unstandardized estimates are the effect of a change in one variable on the other in absolute terms and are equivalent to the slope in a regression model (Lamb *et al.* 2011).

Variable	Unstandardized path coefficient ± Standard Error	*Z* value	*P* value	Standardized path coefficient
Rangeland Health				
Litter	0.040 ± 0.009	4.44	0.001	0.470
Federal	10.150 ± 3.248	3.13	0.002	0.272
Provincial	-7.078 ± 3.786	-1.87	0.062	-0.139
Chestnut-collared longspur				
Litter	-0.006 ± 0.002	-3.00	0.001	-0.369
Vegetation Volume	-0.086 ± 0.025	-3.44	0.001	-0.250
McCown's longspur				
Litter	-0.002 ± 0.001	-2.00	0.015	-0.159
Vegetation Volume	-0.062 ± 0.018	-3.44	0.001	-0.292
Attitude	-1.878 ± 0.613	-3.06	0.002	-0.328
Sprague's pipit				
Rangeland Health	-0.030 ± 0.009	-3.33	0.001	-0.242
Vegetation Volume	0.096 ± 0.016	6.00	0.001	0.444
Litter	0.004 ± 0.001	4.00	0.001	0.365
Size	0.644 ± 0.250	2.58	0.010	0.160
Federal	0.998 ± 0.577	1.73	0.083	0.215
Provincial	1.166 ± 0.424	2.75	0.006	0.184

4.4 Discussion

My analysis identified management jurisdiction and size of land holdings as significant social drivers of rangeland health and abundance of Sprague's pipit. Federally managed lands provided habitat with higher levels of rangeland health and a greater abundance of Sprague's pipit than provincial or private lands. I postulate that this may be due to the conflict between economic growth and biodiversity that differentiates public and private management (Czech *et al.* 2000). Sprague's pipit prefer habitat characteristic of light to moderately grazed rangeland with increased litter and vegetation volume (Knopf 1996). Higher levels of rangeland health on federal lands imply a management approach based on light to moderate grazing pressure that does not emphasize economic returns from grazing; this creates habitat suitable for Sprague's pipit. A greater emphasis on economic returns from livestock grazing on private lands may increase grazing pressure, thereby creating less suitable habitat for Sprague's pipit.

Sprague's pipit, whose abundance was greatest on habitat managed by managers of large land holdings (i.e. >2590 ha), is considered area-sensitive; the probability of occurrence for this species increases with increasing patch size of native prairie (Davis 2004). Large pastures, such as those found on publicly managed lands in the Milk River watershed, represent some of the largest continuous tracts of native prairie remaining in North America (Noble and Kulshreshtha 2007). These large pastures may provide more suitable habitat for a greater number of Sprague's pipits to establish breeding territories compared to smaller pastures found on some private lands. Beyond large publicly managed pastures, the majority of Canadian grasslands are highly fragmented by agricultural production, energy development and human settlements (White *et al.* 2000). Maintaining large patches of native prairie for area sensitive species, such as Sprague's pipit, and engaging public and private managers of large land holdings in prairie conservation should be considered priorities if imperilled grassland species are to be protected.

Another noteworthy finding of my study is that McCown's longspur abundance was greater in habitat characterized by negative manager attitudes towards species at risk. Attitudes towards wildlife can influence a manager's willingness to participate in voluntary stewardship and their approach to rangeland management (Henderson *et al.* 2014). McCown's longspur prefer habitat with low litter and vegetation volume characteristic of increased grazing pressure

(With 1994). As stated earlier, if negative attitudes towards species at risk correlate with a management approach that emphasizes short term economic gain rather than habitat protection, then it is plausible that negative attitudes towards species at risk could result in habitat suitable for McCown's longspur. However, since the mechanism by which attitudes influence McCown's longspur was not captured by the habitat variables included in my model, this requires further investigation.

The bird-habitat relationships I detected support those of previous studies and further demonstrate the distinct preferences grassland birds have for vegetation structure shaped by grazing (Fisher and Davis 2010; Henderson and Davis 2014). Compared to Sprague's pipit, Chestnut-collared and McCown's longspur abundance increased with decreasing litter and vegetation volume and are typically associated with moderate to heavy grazing (With 1994; Davis *et al.* 1999; Henderson and Davis 2014). Since relationships between bird abundance and vegetation features varied by species, my results suggest that a mosaic of habitats varying in vegetation structure is necessary for the conservation and recovery of grassland birds (Samson and Knopf 1996; Fuhlendorf *et al.* 2006; Askins *et al.* 2007).

My study demonstrates how integrating social and ecological data can provide insights into rangeland management drivers of habitat suitability and species abundance for species at risk. Interdisciplinary research is fraught with challenges and barriers, including lack of an efficient, coherent research methodology that integrates data from multiple disciplines (Brandt *et al.* 2013; Roy *et al.* 2013). Along with the use of multiple regression, geographic information systems, and systems analysis (Liu *et al.*, 1999; Brewer *et al.* 2012), structural equation models offer a reproducible method to efficiently integrate social and ecological data. Structural equation models also enable social factors, often considered at the end of natural science based conservation research, to be integrated at the outset of a research project (Fox *et al.* 2006). My use of structural equation modelling demonstrates how social factors, such as management jurisdiction, attitudes towards species at risk, and size of land holdings, influence species abundance both directly and indirectly via their influence on habitat. Only by understanding complex links between social and ecological drivers of species abundance can we potentially curb habitat loss and manage remaining habitat to aid in the recovery of species at risk.

Examining the abundance of multiple species at risk in an interdisciplinary model revealed management implications that might not have surfaced with disciplinary, univariate analysis. Of the social variables I identified as significant to bird abundance, management jurisdiction has the most tangible implications for grassland bird conservation. Although federally managed lands provided more suitable habitat for Sprague's pipit, the lack of relationship between management jurisdiction and either longspur species suggests that longspurs find equally suitable habitat on federal, provincial and privately managed lands. There is a growing body of literature that suggests livestock production is compatible, if not necessary, for the conservation of North American grassland birds (Derner *et al.* 2009). However, although livestock grazing performs an important ecosystem function, grassland species do not respond uniformly to grazing pressure (Knopf 1996). Creating suitable habitat for multiple species requires an understanding of how management jurisdictions are geographically represented across the landscape and how they differ in their grazing management approach (i.e., intensity, duration, frequency). In my study region, a balance of public and private management based on a variety of values (i.e., rangeland health, economic returns) and grazing strategies (i.e., light to moderate to heavy grazing pressure) may provide native prairie habitat for a wider suite of species than one management approach alone. Conservation of the grassland bird community may be best supported by a patchwork of public and private management which, through variation in land holding size and grazing pressure, could create a mosaic of vegetation structure and habitat conditions.

I advocate for a "mosaic by design" rather than a "mosaic by default" approach to native prairie management. Since public and private managers vary in the motivations, pressures and mandates that underlie their management approach, creating such a mosaic is challenging. Public lands are often managed to represent a wider set of societal values than those solely associated with economic return (Platt 2004). The public lands sampled in my study included a national park (i.e., Grasslands National Park) and federal and provincial community pastures. Maintaining and restoring ecological integrity is a primary goal of management in Grasslands

National Park where cattle and bison grazing are used to restore heterogeneity in vegetation structure (Parks Canada 2010). In addition to providing livestock grazing and breeding services for their patrons, federal and provincial community pastures are also managed to maintain ecological integrity. Federal community pastures aim to “manage a productive, bio-diverse rangeland and to promote environmentally responsible land use and practices” (Agriculture and Agri-Food Canada 2007) and provincial pastures “promote environmental and agricultural sustainability of marginal Crown lands through good range land planning and forage management” (Saskatchewan Agriculture 2013). By managing for ecological integrity, albeit to differing degrees, the public lands included in my study represent societal values for ecosystem goods and services (e.g., high rangeland health and habitat for species at risk) beyond economic gain (Havstad *et al.* 2007). While some private managers in my study region consider sustainability in their grazing management, most strive to maintain an economically viable ranching operation which may not include providing habitat for species at risk (Henderson *et al.* 2014).

Creating a mosaic of habitat conditions suitable for multiple species will require a range of strategies suitable for private and public managers. Models for managing habitat for conservation on private lands include conservation covenants, economic incentives, stewardship action plans and conservation agreements (Shogren 2005; Langpap 2006; Rissman *et al*. 2007). The long-standing federal Community Pastures program represents a model of public management that has to date supported habitat protection for species at risk (Kulshreshtha *et al.* 2008). At the time of preparing this manuscript, the federal community pastures which I sampled were in the midst of being dismantled and their ownership transferred to the province of Saskatchewan. It is not yet known if the federal management regimes applied over the last 100 years will be upheld by the provincial government. Hence, the final challenges lie in identifying and coordinating appropriate champions to oversee habitat management at the landscape level and ensuring that individual patches provide conditions that will support a broad range of species.

CHAPTER 5: CONCLUSION- INTEGRATING SOCIAL AND ECOLOGICAL SCIENCE FOR CONSERVATION

5.1 Synopsis

In this thesis, I aimed to identify and explore social and ecological variables of importance to the recovery of grassland songbirds and, more broadly, the conservation of prairie species at risk. With the Milk River watershed of southwestern Saskatchewan as my study area, I employed an interdisciplinary approach to: i)determine the extent to which indices of rangeland health explain variation in grassland songbird abundance for ten grassland bird species, including three species currently listed as *at risk* under Schedule 1 of the Species at Risk Act: Sprague's pipit (*Anthus spragueii*), McCown's longspur (*Rhynocophorus mccownii)* and Chestnut-collared longspur (*Calcarius ornatus*); ii)describe producer characteristics, summarize producer awareness of and attitudes towards species at risk and the Canadian Species at Risk Act and evaluate how characteristics, awareness and attitudes affect producer willingness to engage in voluntary stewardship actions that support species at risk conservation; and iii) explore relationships between social and ecological factors that I postulated were important to abundance of three grassland bird species at risk (Figure 1-1).

In Chapter 2, I demonstrated that the rangeland health index was not a strong predictor of bird abundance. Rather, vegetation structure variables such as litter mass, vegetation volume, and bare ground cover, best explained variation in bird abundance. Vegetation structure variables (i.e., litter and vegetation volume) were present in top-ranking models for eight species and solely comprised top-ranking models for Baird's sparrow *(Ammodramus bairdii)*, Chestnut-collared longspur, Horned lark (*Eremophila alpestris*), McCown's longspur, and Savannah sparrow (*Passerculus sandwichensis*). Structural heterogeneity variables were present in top-ranked models for Grasshopper sparrow (*Ammodramus savannarum*), Horned lark (*Eremophila alpestris*), and Western Meadowlark (*Sturnella neglecta*). Plant composition variables solely comprised top-ranking models for Clay-colored sparrow (*Spizella pallida*) and were present in top-ranked models for Grasshopper sparrow and Vesper sparrow (*Pooecetes gramineus*). Although the rangeland health index received little support as a predictor of bird abundance, I recommend that vegetation structure components of the index be used to communicate grazing management guidelines that maintain grassland bird habitat with livestock producers.

In Chapter 3, I described producer characteristics, attitudes and awareness of species at risk and evaluated how these factors influence willingness to protect species at risk through voluntary stewardship. I found minor variation in producer characteristics in the study region. The majority were male, 3rd generation producers who managed at least 1536 ha of rangeland and learned to manage their lands by experience. While the majority of producers were aware of species at risk and the Species at Risk Act, many were unfamiliar with details of the SARA or the species listing process. Despite this, some relayed detailed information about habitat use, population changes or behavioural observations for species at risk on their lands. Most producers held positive attitudes towards species at risk but would only consider supporting species at risk if it meant that there were no changes to their economic well-being or rangeland management approach. Four attitude themes emerged from my qualitative analysis of interview data: "producers as stewards", "economic risk", "lack of trust and communication", and "do not disturb". First, producers believed that prairie wildlife species, including those considered at risk, would not be in the region if it were not for the stewardship efforts of producers over the last 100 years. Second, they identified their ranching operation as a business and would support species at risk recovery through a stewardship program or new management strategy only under conditions that did not impart increased economic risk. Third, producers relayed a lack of trust and communication with government on issues of species at risk. Finally, producers expressed fear that sharing information about species at risk on their lands would lead to "more encroachment of unwanted people"; they said they would like to be left alone to manage their lands the way they choose. My cluster analysis identified producers aged 60-69 and those who had ranched for more than 50 years as less willing to support species at risk than producers in other categories. Younger producers with higher levels of formal education also displayed greater awareness and positive attitudes towards species at risk. Hence, they were more willing to engage in voluntary stewardship activities to support species at risk. My findings suggest that conserving species at risk on private ranch lands through voluntary stewardship under the SARA is feasible, given certain conditions: financial stability for ranching operations; rewards for sound habitat management; and improved trust between producers and government agencies.

In Chapter 4, I explored relationships between social and ecological factors that I postulated were important to bird abundance for my three focal bird species: Sprague's pipit, McCown's longspur, and Chestnut-collared longspur. I used a structural equation modelling approach to identify social factors, such as management jurisdiction, attitudes and size of land holdings, and ecological factors, such as rangeland health, litter and vegetation volume, as important to achieving grassland bird conservation. Bird species did not respond uniformly to social or ecological variables. Sprague's pipit abundance was better supported by lands managed federally than those under provincial or private management. Relationships between bird abundance and vegetation structure support those identified in Chapter 2. Sprague's pipit abundance was greatest in habitat managed by managers of large land holdings (>2590 ha) and in habitat with increasing litter ($kg \cdot ha^{-1}$), and vegetation volume (cm^3). Habitat with decreasing litter and vegetation volume had the highest abundance of Chestnut-collared and McCown's longspurs. Interestingly, McCown's longspur abundance was also greater in habitat characterised by managers who held negative attitudes towards species at risk. Finally, I demonstrated how the integration of social and ecological data can reveal options for achieving conservation goals.

Overall, my findings suggest that conservation of the grassland bird community may be best supported by a patchwork of grazing patterns currently represented by public and private management which could, through variation in grazing pressure, create a mosaic of vegetation structure and habitat conditions. Designing such a mosaic intentionally will pose a significant challenge in this region where habitat, land ownership and government jurisdiction are fragmented.

5.2 Contributions and Significance

Together, the manuscripts I present in this thesis contribute to the theory, methodology and management considerations surrounding habitat conservation for prairie species at risk on private and public lands.

Theoretically, my findings contribute knowledge related to options for engaging private land managers in species at risk conservation, the role of livestock grazing in grassland bird conservation and grassland bird habitat selection. Although private land management poses challenges for conservation (Knight 1999), results presented in Chapter 3 indicate that private land managers in my study region would support species at risk recovery under the aforementioned conditions. Engaging private land managers may be best achieved by rewarding ranchers for sound rangeland management, providing economic incentives to protect habitat and using social networks in ranching communities. Livestock grazing has been posited as a tool to create or enhance habitat for declining grassland species (Derner *et al.* 2009). The combined findings of Chapters 2 and 3 suggest that, given the right conditions and grazing guidelines, public and private lands supporting livestock production could provide important habitat for species at risk. Finally, my findings uphold the theory that vegetation structure is an important driver of grassland bird habitat selection (Fisher and Davis 2010) and that a mosaic of vegetation structure across the landscape is necessary for the conservation of the grassland bird community as a whole (Fuhlendorf *et al.* 2006).

In terms of methodology, this thesis presents a framework for interdisciplinary study design, data collection and analysis. Although the methods I used to collect data are well established in the fields of ornithology, plant ecology and social science, I present a novel approach to integrate these methods in a study design that samples both social and ecological data from a single given plot (Figures 1-1, A-1). Finally, previous conservation studies have combined social and ecological data with a structural equation model (Mora 2008), however, none to date have done so using primary data or abundance of imperilled species (Figure 4-1).

Finally, this thesis contributes more broadly to the management and conservation of prairie species at risk in general. First, my findings emphasize the importance of the local social context to engaging private agricultural producers in voluntary stewardship (Reed *et al.* 2013). The low levels of producer knowledge of details of the SARA and lack of trust between government agencies and producers that I detected suggest that further consultation and collaboration are needed. These findings, combined with producer characteristics and attitudes that I summarize, have been directly applied to a joint federal and provincial multi-species recovery planning effort called "South of the Divide" that is currently underway in the Milk River watershed. The abundance data for Sprague's pipit collected for this thesis has also been used in modelling critical habitat for this species. As reported elsewhere, my findings identify the wide variation in habitat conditions that is preferred by my three focal grassland bird species at risk. Thus, this thesis highlights the importance of managing grassland to create diversity in vegetation structure that is important for conservation of the grassland bird community as a whole (Knopf 1996).

Finally, the integration and synthesis of social and ecological concepts, methodologies and data presented in this thesis contributes more broadly to the emerging field of interdisciplinary research for biological conservation. By overcoming challenges of interdisciplinary research described below, I contribute to a small, but growing, body of research that integrates social and natural science research at the human-environment interface for conservation (Roy *et al*. 2013). I also demonstrate that graduate students can be trained as interdisciplinary scientists familiar with tools from both social and ecological science without compromising disciplinary expertise or interdisciplinary integration (Moslemi *et al.* 2009; Roy *et al.* 2013). Such training allows for disciplines to be effectively integrated at the outset of research (Fox *et al.* 2006) and creates a league of scientists capable of independently tackling complex issues of conservation biology, rather than relying on the cooperation of scientists from a wide array of disciplines (Roy *et al.* 2013).

5.3 Interdisciplinary Approach – Opportunities and Limitations

The interdisciplinary research presented in this thesis posed several personal, interpersonal and practical challenges and provided me with excellent opportunities for personal and academic growth. First, I was faced with the personal dilemma of gaining sufficient disciplinary knowledge to carry out this work without becoming a specialist in each contributing discipline (Gober 2004; Morse *et al.* 2007; Roy *et al.* 2013). As a trained biologist, I was required to increase my “intellectual agility” (Roy *et al.* 2013) by learning social science theory and methodology. In addition to honing an appropriate level of disciplinary depth, I needed to synthesize concepts from several disciplines and communicate them effectively with members of my supervisory team who hailed from a variety of disciplines. Members of my supervisory committee had faith in the theory and methods of each other’s disciplines and thus provided a strong framework for me to carry out this research (Roy *et al.* 2013). As much as it was a challenge to find the appropriate disciplinary depth, I gained an appreciation for the perspectives, theory and methods of different disciplines. I learned how to frame and visualise an interdisciplinary research problem (Figure 1-1), became proficient in data collection and analytical tools from various disciplines (e.g., R, NVivo, SEM), and published in both social and natural science journals. Finally, in writing across natural and social sciences, I learned new ways of synthesizing interdisciplinary information (Gober 2004).

5.4 Future Research

The findings presented in this thesis provide a small contribution to our understanding of what underlies stewardship for wildlife and how land-use influences habitat for prairie species at risk. Much social, ecological and interdisciplinary research is needed to reverse recent trends in grassland species endangerment.

This thesis prompts three possible areas of social science research. First, for the rangeland health index to be an effective communication tool for biologists and rangeland managers, research is needed to determine the extent to which indices of rangeland health are currently used by rangeland managers on private and public lands and whether using such measures would be feasible for private livestock producers. Second, conservation planners need to understand how changes in private and public land management in the region will affect habitat for species at risk. During interviews, private livestock producers often expressed an uncertainty about the future of management on their lands. The high levels of economic risk associated with livestock production and dwindling rural populations may lead to few family-owned ranches being taken over by the next generation. Also, at the time of preparing this thesis, the federal community pastures which I sampled were in the midst of being dismantled and their ownership transferred to the province of Saskatchewan. It is not yet known if the federal management regimes that created suitable habitat for grassland species will be upheld by the provincial government. Future research could examine how changes in private and public management impact habitat security for species at risk. Finally, Chapter 3 of this thesis identified social networks as a potentially important influence of producer uptake of voluntary stewardship programs. Social networks have been identified as significant to livestock producer involvement in conservation activities on rangelands in Florida (Didier and Brunson 2004) and Utah (Wilcox *et al.* 2012), and would therefore be a useful future research focus in my study region.

In terms of grassland wildlife ecology, the next step is to examine whether a revised rangeland health index is a feasible tool for assessing grassland biodiversity. To start, one could examine how vegetation measures related to rangeland health and those traditionally collected for ornithological studies of bird-habitat relationships compare in their ability to explain variation in bird abundance. Understanding whether and how these measures are related may help to refine the rangeland health index as a tool to assess habitat suitability for grassland birds. Future research could also refine the rangeland health index to include greater emphasis on vegetation structure and then examine relationships between the refined index and bird abundance or overall grassland biodiversity. Future research should also investigate the extent to which grassland bird breeding success relates to rangeland health. Answers to such questions would enable identification of pastures as source or sink habitat, thereby improving knowledge of habitat quality and the ability to prioritize habitat protection.

Finally, this thesis emphasizes the importance of interdisciplinary research to the conservation of prairie wildlife. None of the social and ecological variables I identify in Chapter 4 as important to bird abundance are static over time, which raises many questions: Do changes in social factors such as governance and policy, livestock market prices or rural depopulation influence the quality and quantity of habitat for prairie wildlife? If so, how? How do ecological factors like drought and fire interact to affect the availability of habitat? How might the quality or availability of habitat for prairie wildlife change with a changing climate? How do social (e.g., livestock market prices) and ecological factors (e.g., drought) interact to influence habitat availability for prairie wildlife? These questions frame an ambitious research program best carried out with long-term monitoring and an interdisciplinary research team.

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APPENDIX A: SUPPLEMENTAL INFORMATION FOR CHAPTER 2

Figure A-1. Quarter section sampling scheme depicting approximate placement of point count locations (solid circles) and radius (100 m), vegetation sampling line transects (minimum 300 m; dashed lines) and Daubenmire frames (boxes).

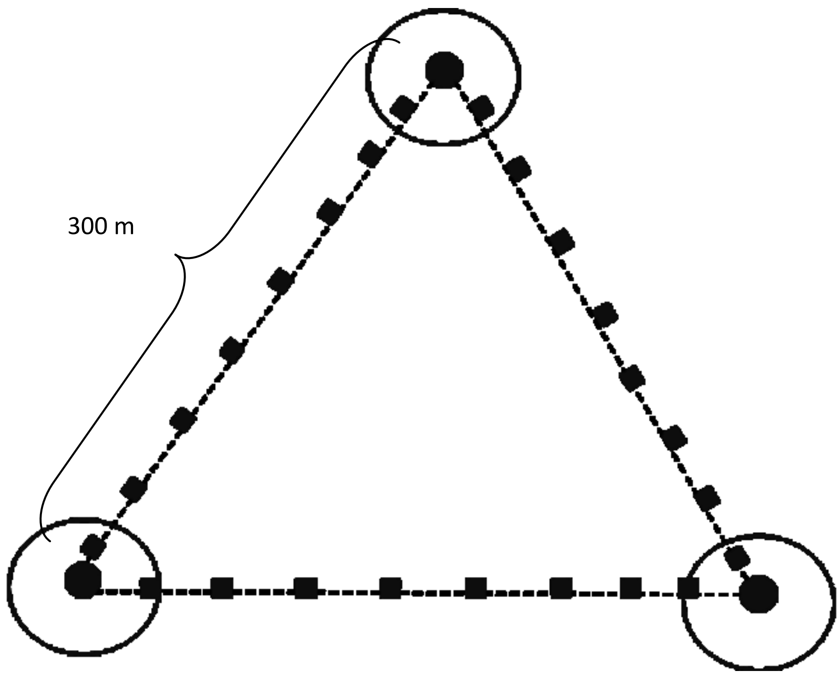
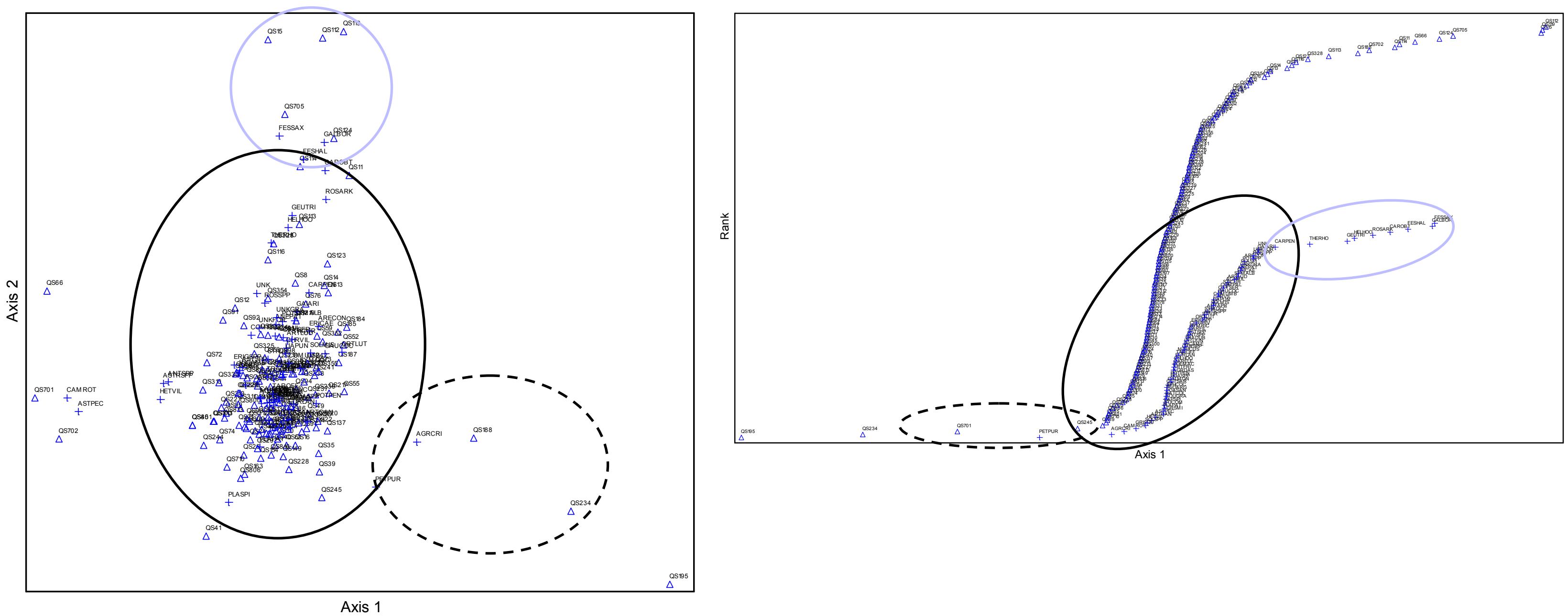


Figure A-2. Plot of **A)** two-dimensional and **B)** one-dimensional Non-metric Multidimensional Scaling of plant species composition depicting plant community structure across study sample units. In both, 74 plant species and quarter sections are partitioned into those typical of the Dry Mixed or Mixed Grassland ecoregion (black solid), Cypress Upland Grassland ecoregion (grey) or native grassland dominated by invasive species (black dashed).



A) **B)**

Figure A-3. Cleveland dot-plots displaying high frequency of zeros in bird abundance data for Baird's sparrow (BAIS), Chestnut-collared longspur (CCLO), Clay-colored sparrow (CCSP), Grasshopper sparrow (GRSP), Horned lark (HOLA), McCown's longspur (MCLO), Savannah sparrow (SAVS), Sprague's pipit (SPPI), Vesper sparrow (VESP) and Western Meadowlark (WEME). The x axis represents the number of birds detected for each focal species. The y axis is by default the order of observations in the data frame.

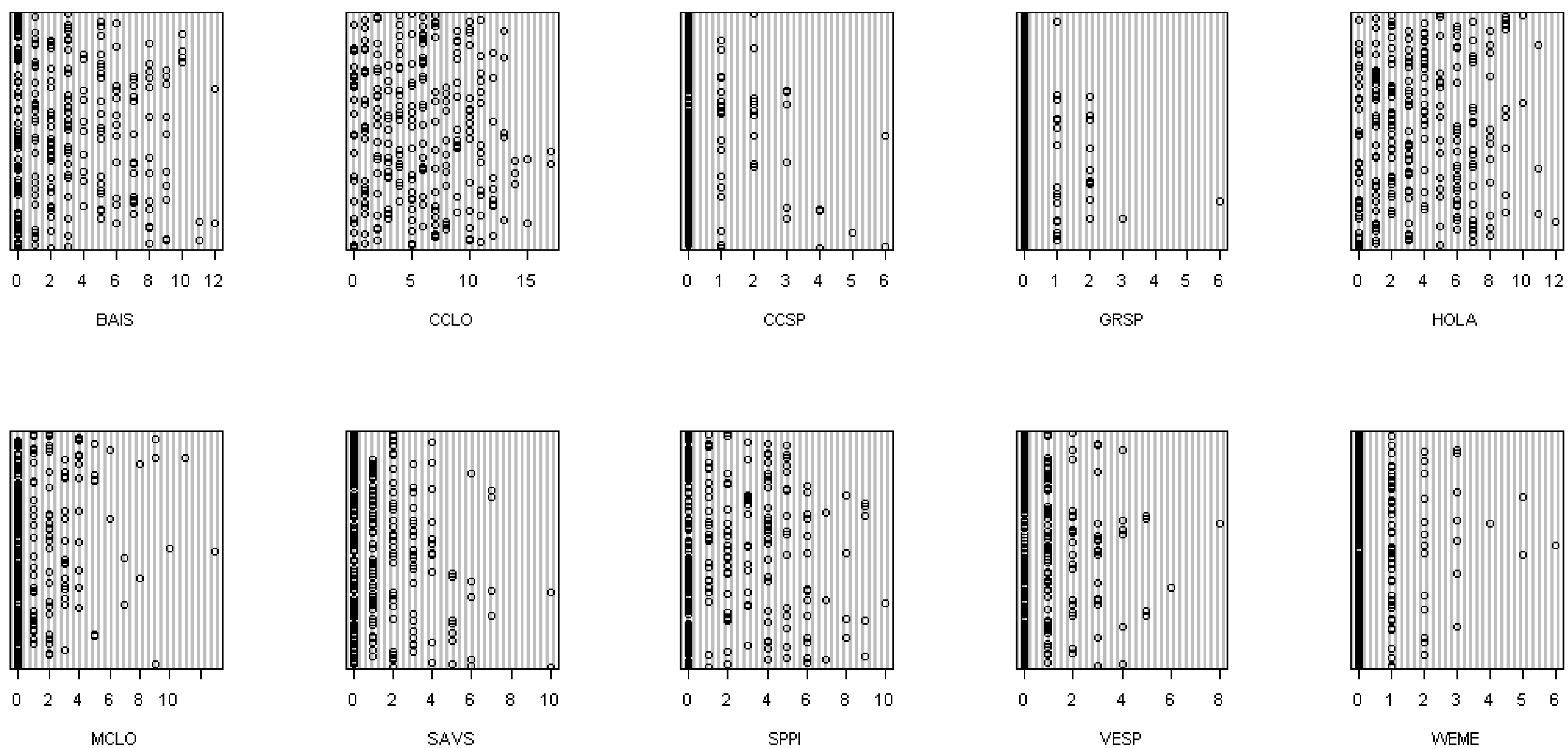
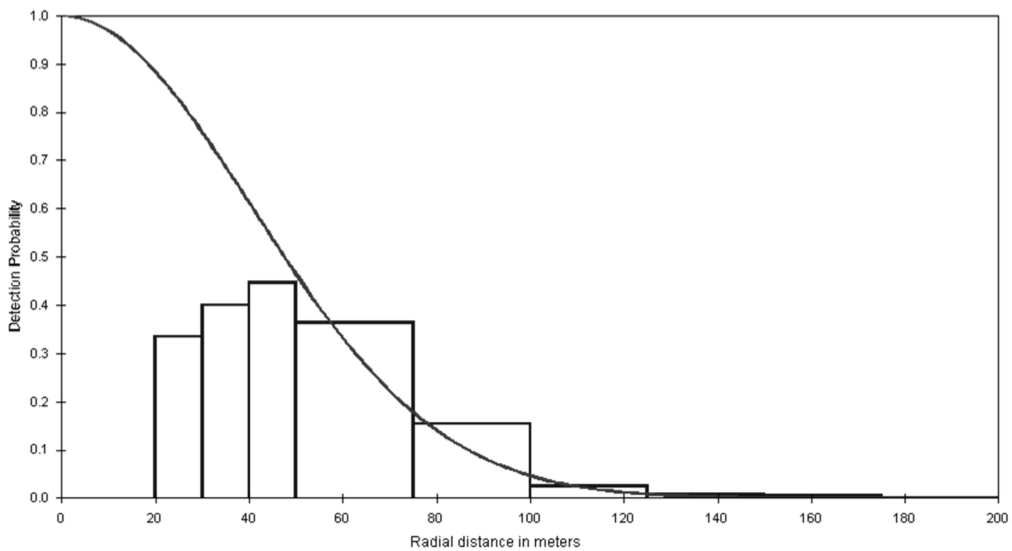
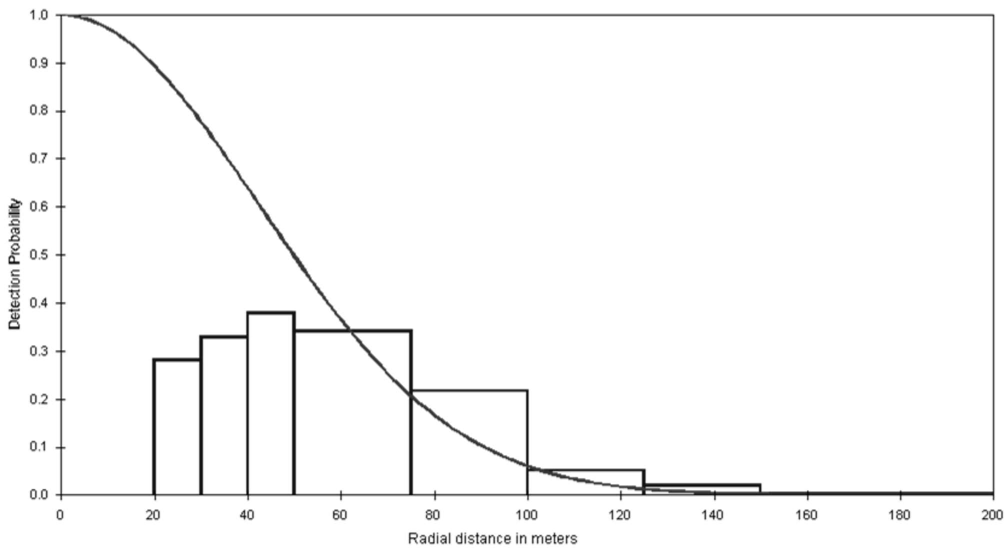


Figure A-4. Example histograms of grassland songbird detections as a function of distance (m) and the estimated detection probability function (solid line) from point count surveys conducted in 2009 and 2010 in the Milk River watershed of southwestern Saskatchewan, Canada. All models indicate that the detection probability is <1.0 within the first distance interval, thereby violating the first assumption of distance analysis. Species presented are **A)** Chestnut-collared longspur, **B)** McCown's longspur and **C)** Sprague's pipit.



A)


Detection Probability
Radial distance in meters
1.0
0.9
0.8
0.7
0.6
0.5
0.4
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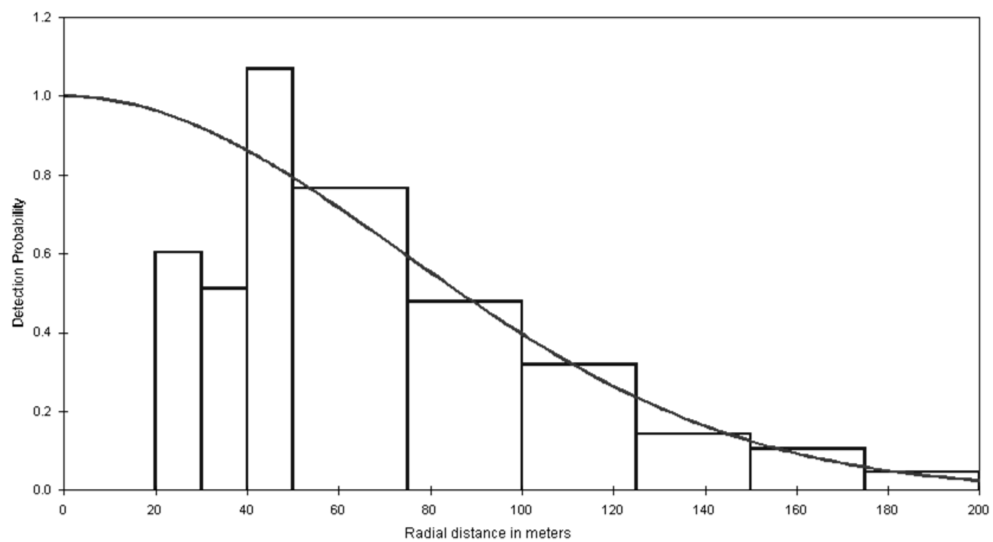
100
120
140
160
180
200

B)


Detection Probability
Radial distance in meters
1.2
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120
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160
180
200

C)

Table A-1. RMARK model selection results showing best performing models of detection probability for 10 grassland bird species in southwestern Saskatchewan, where min= minutes since sunrise, rh= rangeland health score, lit= litter mass (kg·ha-1), cloud= % cloud cover, obsvr= observer and robel= vegetation volume (cm^3).

Species	Best fit model	K	AICc	w_i	Estimate of Detection Probability p(~)	SE
Baird's sparrow	p(~Time)	2	1361.47	0.37	-0.39	0.20
Chestnut-collared longspur	p(~1 + min)	2	8026.29	0.30	0.00	0.00
Clay-colored longspur	p(~1)	1	165.26	0.20	-0.09	0.31
Grasshopper sparrow	p(~1)	1	72.17	0.15	-2.32	2.42
Horned lark	p(~1 + rh)	2	3042.92	0.29	0.02	0.01
McCown's longspur	p(~1)	1	553.09	0.12	-1.21	0.34
Savannah sparrow	p(~1 + lit)	2	727.11	0.46	0.00	0.00
Sprague's pipit	p(~1 + cloud + obsvr)	3	662.72	0.21	cloud: 0.01 obsvr: -0.37	cloud: 0.004 obsvr: 0.24
Vesper sparrow	p(~1 + min + obsvr)	3	394.88	0.14	min: 0.01 obsvr: -1.01	min: 0.004 obsvr: 0.88
Western meadowlark	p(~1 + robel)	2	180.83	0.89	-0.30	0.12

Table A-2. Rangeland health parameter estimates (β^1) and associated standard errors (SE) for 10 grassland bird species in southwestern Saskatchewan, showing AICc values for the null model of no effect, log likelihood values (log(L)) and AICc values for rangeland health and ΔAICc values (Δi) relating rangeland health AICc values to those of best models from Table 2. Parameter estimates whose 85% confidence intervals include zero appear in *italics*. The number of model parameters (K) is three throughout.

SPECIES	NULL AIC_c	log(L)	AICc	Δ_i	β^1	SE
Baird's sparrow	1136.3	-535.17	1082.7	15.89	*0.13*	*0.34*
Chestnut-collared longspur	1378.3	-664.05	1340.4	20.02	-0.41	0.28
Clay-colored sparrow	349.7	-167.32	344.9	7.75	*0.50*	*1.06*
Grasshopper sparrow	241.5	-112.84	235.5	7.56	*1.85*	*1.40*
Horned lark	1188.8	-578.24	1168.8	28.11	*-0.15*	*0.31*
McCown's longspur	755.1	-346.05	704.4	21.22	-0.95	0.60
Savannah sparrow	822.4	-387.77	787.9	20.55	1.36	0.46
Sprague's pipit	927.5	-439.37	891.1	11.71	*-0.22*	*0.42*
Vesper sparrow	628.4	-310.13	628.4	12.14	*-0.91*	*0.64*
Western meadowlark	439.9	-215.72	441.7	3.82	*0.43*	*0.75*

APPENDIX B: SUPPLEMENTAL INFORMATION FOR CHAPTER 3

Interview Questions

History, Heritage and Culture

1. How long have you lived in the area?

2. Do you ranch or farm or both?

3. How long have you been doing this?

4. Have previous generations of your family members also been involved in your operation?

    a. If (yes), for how long?

Land Management

5. How much land do you manage?

<1 section   1-5 sections   6-10 sections   > 10 sections

    a. How much of your land is native prairie?

<1 section   1-5 sections   6-10 sections   > 10 sections

6. How much land do you farm? Ranch?

    <1 section   1-5 sections   6-10 sections   > 10 sections

7. What are the most important strategies you use to manage your native prairie?

8. How did you learn to manage your land? Why?

    a. So would you say it is through education or experience, or both?

9. Who makes land management decisions in your family? Where does your information come from for making management decisions? Which are most important? Why?

    family   media   neighbours   workshops

    internet   education   experience   newspaper

Species at Risk

10. What kinds of wildlife use your land?

11. Why is wildlife important to you? Why?

Hunting for food | Hunting for sport | Entertainment

Inspirational/Something to appreciate | Indicator of healthy landscape | Economic

12. Have you heard the phrase 'species at risk'?

13. What does the term, 'species at risk' mean to you?

14. Are you aware that Canada has a *Species at Risk Act*?

    a. Where does the Species at Risk Act apply legally?

15. What are three species at risk in the grasslands near here?

16. Do you think they should be considered species 'at risk'? Why?

17. Do you think that the government is legally obligated to conserve species at risk?

18. Do you know how species at risk are determined?

    a. If yes, who determines whether they are at risk?

19. Nowadays, where does your information about species at risk come from? Why?

    family | television/radio | neighbours | workshops/meetings

    gov't representatives | internet | newspaper | mail-outs

20. If you had species at risk on your property, would you share this information? Why?

21. Would you be willing to adopt a new grazing strategy if it meant supporting a species at risk?

    i. If yes, have you used any? Why?

    ii. If no, why not?

22. How far would you go to support a species at risk? For example would you implement a rotational grazing system? Or would you put up a fence to protect species at risk habitat?

23. What incentives, if any, might be useful for you to adopt new management practices?

economic access to information technical assistance

24. How would you like to receive information about new incentive programs or management practices that will help support species at risk?

Mail phone in person workshop/meeting email

25. In your opinion, what are the biggest limits to adoption of new management practices for your operation?

information money/capital technology labour

26. Have you every participated in a government stewardship program? If so, which one?

   a. If not, would you be involved in a government stewardship program? How?

27. If there was an opportunity, would you like to be included in SAR recovery planning in your region? If so, how?

28. Do you think that Land Managers/ranchers are doing a good job of maintaining species at risk on their lands?

29. Do you think that the government is doing a good job of maintaining species at risk on the prairies?

30. Do you think the government should be involved in working with land managers to maintain species at risk on the prairies? Why or why not? If so, how?

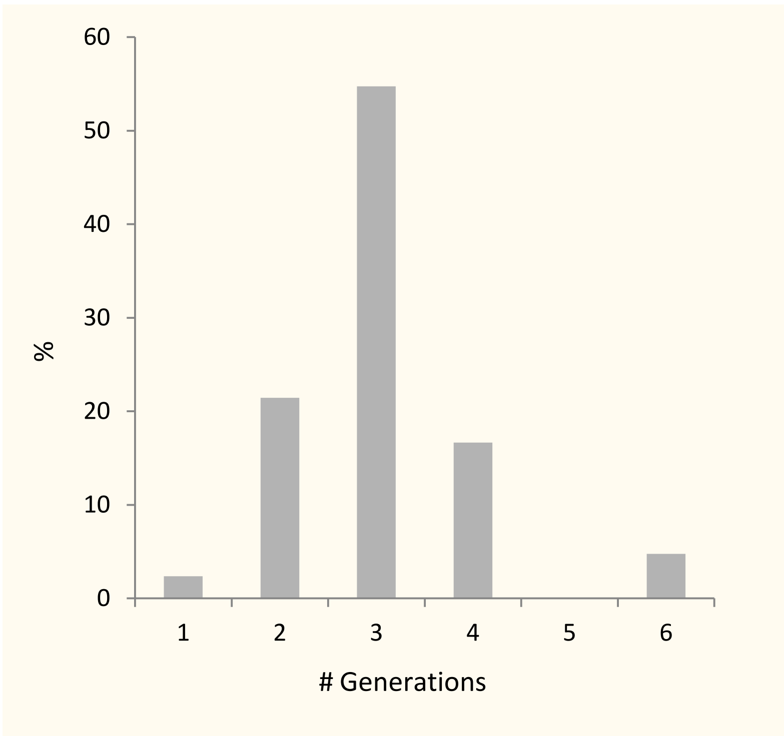
31. What do you think needs to happen to ensure the long term viability of people and wildlife on the prairies?

In Closing

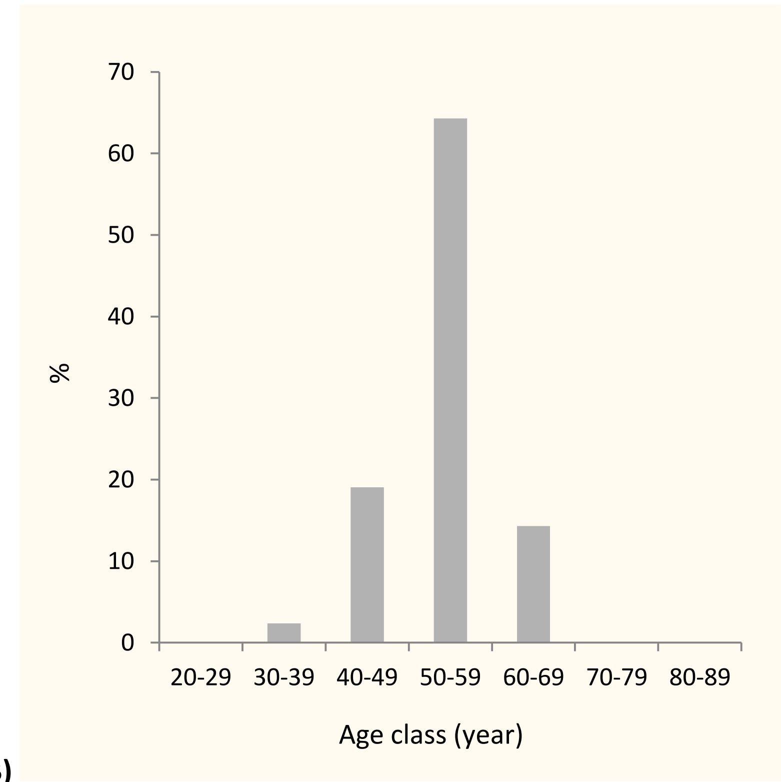
32. Do you have any general comments about the things we’ve talked about here?

33. Do you want to go back over anything?

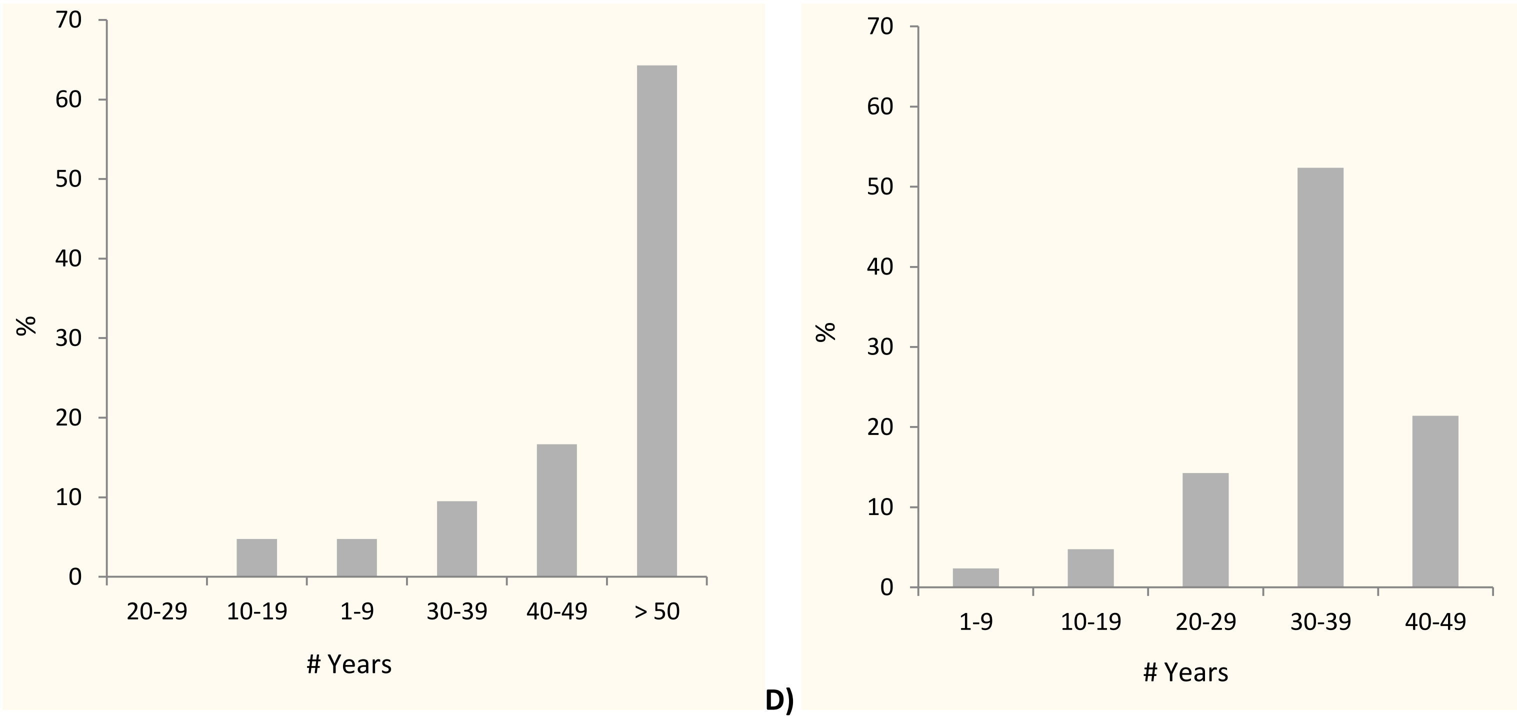
Figure B-1. Characteristics of producers including **A)** the number of generations managing the ranch, **B)** producer age, **C)** duration lived in the region, **D)** duration of producer's lifetime spent ranching and **E)** current land holding size. Values are expressed as percentages of total number of producers.



A)



B)


70
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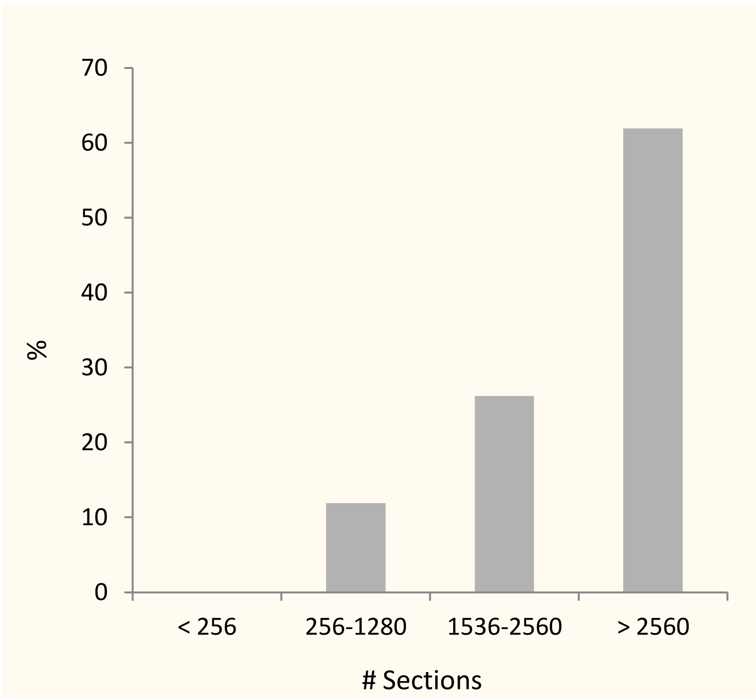
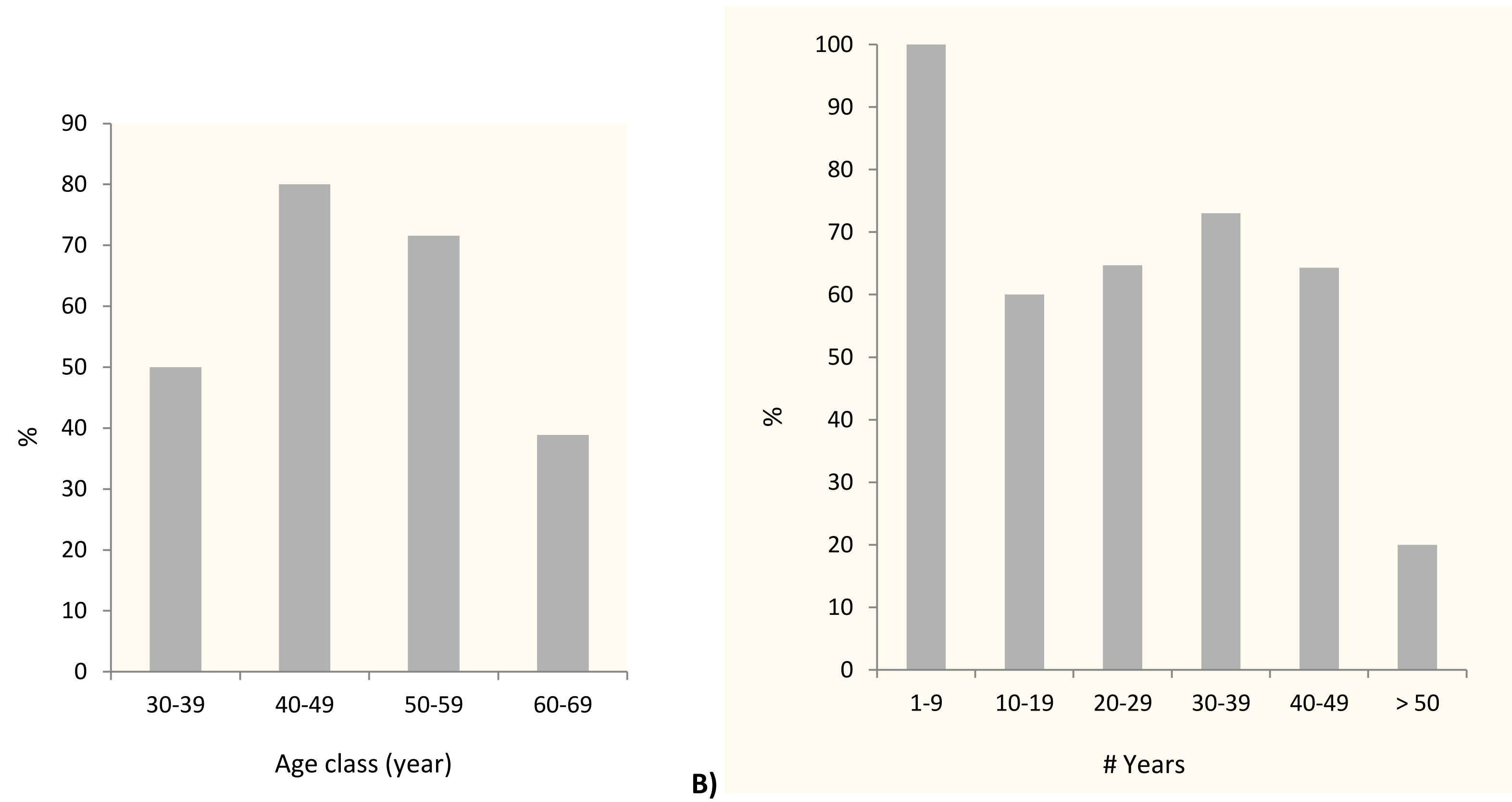
40-49
Years

C) **D)**



E)

Figure B-2. Generalized willingness to support species at risk recovery for **A)** producers age and **B)** duration spent ranching. Percentages are the proportion of coding references within each year class. Generalized willingness is indicated by willingness to share information about species at risk on their lands, adopt a new management strategy, and participate in species at risk recovery planning.



A) **B)**

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